Environment Ontario Research Report

TREATMENT OF LANDFILL LEACHATE BY SPRAY IRRIGATION
(Muskoka Lakes)

Prepared for:
Ontario Ministry of Environment
R.A.C. Project No. 244-RR

February 1988

UNIVERSITY of GUELPH

TREATMENT OF LANDFILL LEACHATE BY SPRAY IRRIGATION
(Muskoka Lakes)

Research Advisory Committee
Project No. 244-RR

by

R.A. McBride, A.M. Gordon and P.H.Groenevelt
Departments of Land Resource Science and Environmental Biology
University of Guelph
Guelph, Ontario
N1G 2W1

for

Waste Management & Policy and Planning Branches
Ontario Ministry of Environment
135 St. Clair Avenue West
Toronto, Ontario
M4V 1P5

February 1988

DISCLAIMER

"This report has been reviewed by the Research Advisory Committee of the Ministry of the Environment and approved for publication. Approval does not necessarily signify that the contents reflect the views and/or policies of the Ontario Ministry of the Environment nor does mention of trade names or commercial products constitute endorsement or recommendation for use".

UNIVERSITY OF GUELPH
ONTARIO AGRICULTURAL COLLEGE
Department of Land Resource Science

GUELPH, ONTARIO, CANADA · N1G 2W1
Telephone (519) 824-4120

Rerum Cognoscere Causas

October 28, 1987

Ontario Ministry of the Environment
Waste Management Branch
40 St. Clair Avenue West
4th Floor
Toronto, Ontario
M4V 1P5

Attention: Mr. Aki Oda, P. Eng.
Senior Waste Processing Engineer and
Project Liaison Officer

Dear Sir:

Re: Treatment of Landfill Leachate by Spray Irrigation - Muskoka Lakes (Final Report) - M.O.E. Project 244-RR

We respectfully submit our final report on research carried out in 1986 at the Muskoka Lakes MSW landfill site, the only known location in Ontario to have implemented a relatively large scale land treatment/disposal system for leachates. This was a highly interdisciplinary effort which has yielded a number of very significant findings in a research area which has received little attention in Ontario in the past.

Recent provincial government initiatives in the area of energy from waste technology development underscore the need to divert growing pressure on landfilling as the primary means of disposing of domestic wastes in Ontario to other methods of waste management. The inevitability of leachate generation and the environmental hazard it poses, particularly at decommissioned and derelict landfills across Ontario, must be dealt with by Municipalities in a simple and cost-effective manner. Slow rate infiltration land treatment represents one such method but much more experience through pilot-scale perturbation studies under different biophysical and climatic conditions is required before its suitability can be fully gauged. This report represents an attempt to resolve some of the difficulties that have been experienced at one such site (Muskoka Lakes) and sets out a research methodology for further investigation at other MSW landfills in Ontario.

We thank you for the opportunity to undertake this most interesting research project.

Sincerely,

R. A. McBride

RAM:mm
Encl.

R. A. McBride
Assistant Professor

ACKNOWLEDGEMENTS

This report is issued to describe an environmentally oriented research project conducted at the University of Guelph and supplemented by a grant from the Research Management Office, Policy and Planning Branch, of the Ontario Ministry of the Environment. All enquiries regarding this project should be directed to Mr. A. Oda, P. Eng. (Waste Management Branch) or to the Research Management Office of the Ontario Ministry of the Environment.

The M.O.E. Scientific Authority and research direction for the project was provided by a steering committee comprised of A. Oda (Liaison Officer), B. Howden, J. Beaver and R. Pearson. The manuscript was reviewed by this committee and A. Scott, also of the M.O.E. Invaluable logistical support and additional background information was also provided by the Corporation of the Township of Muskoka Lakes and its officials. Particular appreciation is extended to Mr. P. Dwyer (Superintendent of Public Works) and Ms. S. Hatch (Mayor).

Drs. K.M. King, T.E. Bates and R.P. Voroney of the Department of Land Resource Science (Univ. of Guelph) contributed greatly to the scientific integrity and interdisciplinary methodology employed in the study.

The research assistants who were responsible for the daily operation of the field and laboratory project components and who assisted in the report preparation were A. Fisken, J. Cuthill, and P. Cureton. The sole graduate research assistant in 1986 was B. O'Neill. Part-time and summer assistants included P. Smith, T. Volk, L. Bober, D. Woll, P. Langille and M. White. Cartography for the report was supervised by D. Irvine and the manuscript was typed by M. Metcalf.

TABLE OF CONTENTS

LIST OF TABLES

Page

LIST OF FIGURES

Page

1.0 Executive Summary

This report describes the details of a research study conducted to solve the problem of municipal solid waste (MSW) leachate management at the Muskoka Lakes landfill site (MOE Certificate of Approval #A510735) located on Lots 21, 22 and 23, Concession I, Medora Ward, Township of Muskoka Lakes, District Municipality of Muskoka. This is the first known location in Ontario to have installed a relatively large-scale slow rate infiltration land treatment system (spray irrigation) for MSW leachate. The project was undertaken to address a number of environmental concerns which had become apparent in the short time that the spray irrigation system had been in operation.

The conclusion from the research conducted in 1986 is that the present method of spray irrigation for leachate management is not a long-term, environmentally sound and economically feasible solution to the Muskoka Lakes sanitary landfill seepage problem. Nearly all of the results from the field tests have shown that soil waterlogging was the underlying cause of adverse environmental impacts on the soil/forest vegetation ecosystem. This was due to iron-induced and irreversible soil morphological changes which have led to evidences of forest decline and localized forest mortality.

Spray irrigation as currently practised on site is not considered an environmentally sound solution for leachate management for the following reasons:

(a) large volumes that must be disposed on a limited land area;

(b) poor distribution of leachate in the forest ecosystem;

(c) soils possess undesirable physical and chemical properties for the effective operation of a spray irrigation system;

(d) soils do not have the capability to retain and attenuate the leachate and its constituents;

(e) existing systems require extensive labour and maintenance costs and therefore may not be considered as a cost effective method to eliminate completely the leachate's eventual emergence in the perennial stream and the North Bay of Lake Muskoka.

The Municipality could improve the future prospects of continuing the spray irrigation for leachate management at this site. The following corrective steps are suggested:

(a) installation of a suitable landfill cover;

(b) aquifer decontamination through dewatering operations at the base of the landfill;

(c) recirculation of leachate through the landfill during the winter months;

(d) establishment of vegetation on the small watershed draining into the present leachate collection trench.

The problem of soil waterlogging was greatly aggravated by poor spray distribution. This was a result of current spray irrigation from a fixed nozzle system operated in a non-overlapping mode. This has clearly demonstrated that if the application rates exceed a critical threshold level, it can cause a general forest decline which could escalate to forest mortality. Table 1.1 attempts to segregate the major causes of vegetative stress and general forest decline into predisposing, inciting and contributing factors.

1.0 Sommaire

Le présent rapport porte sur l'étude effectuée en vue de trouver une solution au problème que posent les produits de la lixiviation des ordures ménagères à la décharge des lacs de Muskoka (certificat d'autorisation du ministère de l'Environnement n^o A510735), située sur les terrains 21, 22 et 23, concession I, quartier de Medora, canton des lacs de Muskoka, municipalité de district de Muskoka. C'est le premier endroit connu en Ontario à avoir mis sur pied un système relativement important de traitement des sols par infiltration lente (arrosage par aspersion) des produits de la lixiviation des ordures ménagères. On a entrepris ce projet pour étudier une série de problèmes environnementaux surgis pendant la brève période d'application de l'arrosage par aspersion.

Des recherches menées en 1986, on a conclu que la méthode actuelle d'arrosage par aspersion, pour la gestion des produits de lixiviation, ne constitue pas une solution valable à long terme ni sur le plan de l'environnement, ni sur celui de la faisabilité économique, en ce qui concerne les problèmes d'infiltration à la décharge contrôlée des lacs de Muskoka. Presque tous les résultats des essais sur place ont prouvé que l'imprégnation d'eau du sol avait des effets néfastes sur l'écosystème sol/forêt. Ils étaient dûs à des modifications morphologiques irréversibles du sol, provoquées par le fer, qui ont entraîné un déclin de la forêt et son dépérissement à certains endroits.

Voici pourquoi l'arrosage par aspersion qu'on pratique actuellement à cet endroit ne peut être considéré comme une solution valable pour la gestion des produits de lixiviation:

(a) le volume d'ordures est trop important par rapport à la superficie du terrain;

(b) les produits de lixiviation se répartissent mal dans l'écosystème forestier;

(c) les sols possèdent des propriétés physiques et chimiques contraires au bon fonctionnement d'un système d'arrosage par aspersion;

(d) les sols sont incapables de retenir les produits de lixiviation et leurs constituants et d'en atténuer les effets;

(e) le système actuel exige beaucoup de main-d'oeuvre et coûte cher en maintenance et ne peut donc être considéré comme une méthode d'un bon rapport rendement-prix pour empêcher totalement les produits de lixiviation de finir par aboutir dans le cours d'eau permanent et la baie nord du lac de Muskoka.

La municipalité pourrait améliorer les possibilités ultérieures de la poursuite de l'arrosage par aspersion pour la gestion des produits de lixiviation à cet endroit. Voici les mesures préconisées à cet effet :

(a) installation d'un recouvrement adéquat de la décharge;

(b) décontamination de la couche aquifère par des travaux d'assèchement, à la base de la décharge;

(c) recirculation des produits de lixiviation à travers la décharge pendant les mois d'hiver;

(d) plantation d'une couverture végétale sur le petit bassin hydrographique dont les eaux s'écoulent dans l'actuel fossé collecteur des produits de lixiviation.

Le problème de l'imprégnation d'eau du sol a été sérieusement aggravé par la mauvaise distribution de l'aspersion, provenant d'un arrosage par aspersion effectué à l'aide d'une buse fixe, fonctionnant par passes ne se chevauchant pas. Tout ceci a prouvé clairement que si les taux d'application dépassent un seuil critique, ce procédé peut entraîner une régression générale de la forêt pouvant éventuellement déboucher sur son dépérissement. Le tableau 1.1 constitue un essai de classification des principales causes des agressions contre la végétation et du déclin général de la forêt en facteurs prédisposants, incitatifs et contributifs.

Table 1.1. Factors responsible for forest decline and mortality at the Muskoka Lakes landfill site.

Predisposing Factors	Inciting Factors	Contributing Factors
· coarse-textured soils shallow to bedrock (deficient water and nutrient availability, inadequate root support)	· poor soil aeration (low soil oxygen for root respiration) caused by waterlogging and application of high BOD/COD wastewater	· insect, disease and microbial infestations in vegetation left vulnerable by irrigation-related damage
· acid precipitation	· micronutrient (Fe, Mn) imbalance and possible toxicity	
	· formation of Fe-indurated soil layer of low permeability to water, air and roots	
	· reduced plant photosynthesis and transpiration caused by excess soil water and foliar staining in the understory	

2.0 Introduction and Study Objectives

2.1 Preface

It is well recognized that landfilling is the predominant method of solid waste disposal in Ontario at this time. Furthermore, due to environmental as well as economic constraints associated with many alternative disposal procedures (e.g. energy from waste [EFW] technology), landfilling is likely to remain an important method of solid waste management for some time to come.

One of the most important and persistent environmental problems associated with municipal sanitary landfills is the pollution risk which results from the generation and subsequent migration of landfill leachate (Chian and Dewalle,1976; Menser et al., 1978). This report documents the research which was conducted from May 1986 to March 1987 at the Muskoka Lakes landfill (M.O.E. file #A510735) under the aegis and with the support of the Ontario Ministry of the Environment (M.O.E. Project 244-RR).

2.2 Study Goal and Objectives

Consistent with the environmental concerns expressed above, an overall study goal was identified as well as a series of more specific objectives. These pertain to the first year of a longer term research plan which would be necessary to fully address the known environmental problems at the Muskoka Lakes site.

Study Goal

To evaluate leachate spray irrigation as a long-term, environmentally-sound and economically feasible solution to the Muskoka Lakes sanitary landfill seepage problem.

Study Objectives:

1) To review the existing research and consultant publications relating to the collection, treatment and land disposal of sanitary landfill leachate. Emphasis should be placed on the practice of spray irrigation in different ecosystems as a form of slow rate infiltration land treatment.

2) To broadly evaluate the surface and subsurface hydrology of the municipally-owned lands in the vicinity of the capped landfill for the purpose of recommending means of reducing the volume of leachate being generated.

3) To investigate the attenuation capacity of the soils within the present spray area with a view to ascertaining the maximum application rates.

4) To broadly evaluate the nature and severity of the stresses imposed on the mixed hardwood forest ecosystem by leachate exposure in ascertaining the capacity of this ecosystem to withstand spray irrigation of leachate over the long-term.

5) To assess appropriate physico-chemical means of leachate pretreatment assuming that phytotoxicity from leachate constituents is a major cause of vegetative stress.

It was recognized at the outset of the study that only objectives 1-3 could be fully investigated within the first year of the broader research plan. The report deals primarily with these findings. Preliminary results pertaining to objectives 4 and 5 are also presented.

2.3 Research Needs

Sufficient rationale exists to support a comprehensive research effort into slow rate infiltration land treatment (spray irrigation) for the management of leachates in the more isolated areas of Ontario. First and foremost is the limited availability of published research information on this disposal method. A large proportion of published work on the topic originates in the United Kingdom, and to a lesser extent in the United States, and does not lend itself to ready interpretation or application under Ontario climatic and biophysical conditions. Second, the spray irrigation area situated to the west and upslope of the perennial stream remains inadequately characterized with respect to soil infiltration rates, hydraulic conductivities, soil water storage capacities and seasonal groundwater table levels. This information, in conjunction with better estimates of evapotranspirational demand, is required to minimize the occurrence of soil saturation, surface runoff and deep drainage during spray operation periods. Third, visibly adverse effects have been observed on the vegetation within the spray areas but the nature and severity of these effects are unknown (i.e. phytotoxicity, direct obstruction of radiation needed for photosynthesis).

2.4 Background

The Muskoka Lakes sanitary landfill site is the first known location in Ontario to have installed a relatively large scale municipal landfill leachate collection/land treatment (spray irrigation) disposal facility. Solid waste landfilling began in April, 1976 and continued until the site was decommissioned in 1980. Shortly thereafter, clay capping material was applied to the landfill surface in an attempt to mitigate leachate generation which had been observed as early as 1978. At the time that landfilling ceased, there were an estimated 1.45×10^4 tonnes of waste emplaced with a density of approximately $0.45\ Mg \cdot m^{-3}$.

Early in the landfill's history, Gartner Lee Associates Ltd. (1978) reported that source removal through excavation and transport to a more environmentally acceptable landfill was the most effective and long-term solution to the leachate problem from several alternatives, although a cost-benefit analysis was not performed to support this view. The report further stated that interim remedial action was required while an appropriate site was selected. One of several alternative solutions falling under the general heading of leachate collection/treatment (i.e. trench infiltration) was adopted for this intervening period. This system required the construction of a collection trench, a leachate pumping system and nearby infiltration trenches (Figure 2.1). The sump pump initially employed proved inadequate as frequent clogging was caused by sediment and leachate sludge (Totten, Sims, Hubicki Assoc. Ltd., 1983). An expanded pump system which was subsequently installed only exacerbated the disposal problem since the infiltration trenches were incapable of handling the volumes applied and overland flow resulted (M.M. Dillon Ltd., 1980).

The lack of success in identifying a suitable long-term disposal site within the municipality for source removal prompted an investigation into spray irrigation of leachate into the adjacent and forested municipally-owned lands. This was viewed as a plausible means of attenuating the contaminants through evapotranspiration, volatilization and accumulation in the soil/vegetation ecosystem (M.M. Dillon Ltd., 1980). This method was further recommended on the grounds of practicability, simplicity, flexibility and cost. Once again, with the exception of the preferred leachate collection/treatment facility, a detailed cost-benefit analysis was not undertaken for the 12 alternative remedial options to support this recommendation.

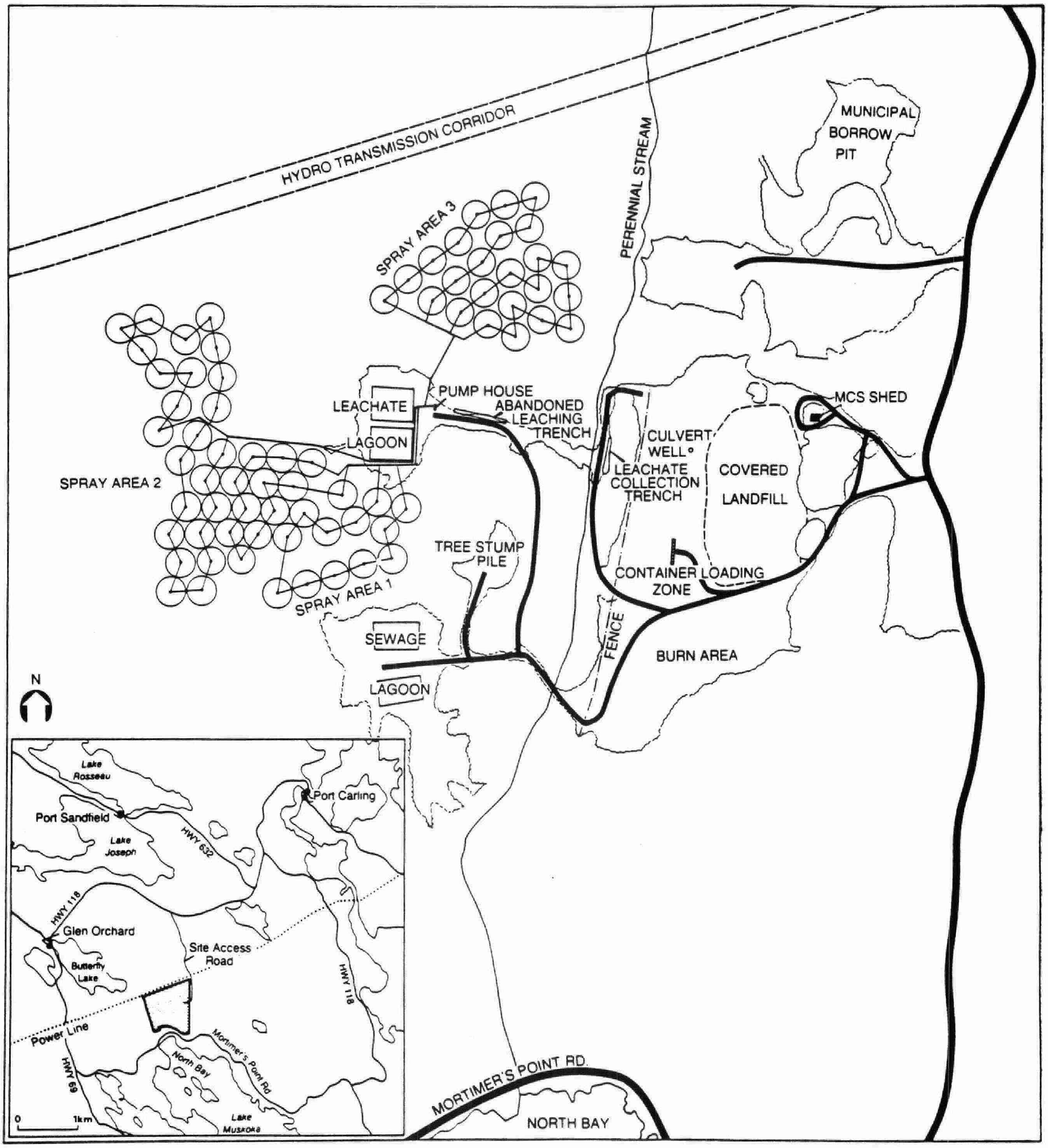


Figure 2.1. The Muskoka Lakes MSW landfill site (scale 1:6,300).

In 1980, a single, portable spray nozzle was installed in the forested area to the north of the septic tank sludge lagoons (Figure 2.1). It was recognized that this moveable nozzle should be relocated periodically to limit soil and groundwater contamination. It became apparent that one nozzle was inadequate and two years later the system was upgraded to a fixed, six-nozzle system. The expanded system showed promise in terms of leachate attenuation but chronic saturation of the sprayed soils resulted in surface ponding and frequent breakouts of overland flow. In 1983, a further study report reiterated that the shallow depth of soils to bedrock (i.e. generally <2m) caused oversaturation and therefore recommended that a larger area be spray irrigated and lower daily volumes be applied (Totten, Sims, Hubicki Assoc. Ltd., 1983). As a result, a larger system consisting of three force mains, each with 24 standpipes fitted with revolving Rain Bird irrigation nozzles, was constructed and commenced operation in July, 1985. The present system consists of a leachate collection trench situated downslope from the capped landfill, two settling lagoons where a limited degree of aeration and microbial pretreatment occurs, and a network of 72 revolving spray nozzles which distributes this partially-treated leachate over three areas totalling approximately 4.3 ha. in a mixed hardwood forest (Figure 2.1).

A research proposal reflecting the immediacy of the problem and the concerns raised by the local Municipal authorities was submitted to the Ontario Ministry of the Environment by the University of Guelph in March, 1986 to study the treatment of landfill leachate by spray irrigation at the Muskoka Lakes site. It had been estimated that, left unabated, a minimum of 2.4×10^7 ℓ of leachate will be produced and collected annually at this site for decades to come. This would exceed the existing system's annual maximum disposal capacity of about 2.0×10^7 ℓ, given recommended spray application rates and the land area available for disposal. Expansion of the current system is virtually impossible due to physiographic constraints within the municipally-owned property boundaries. Furthermore, insufficient published research exists on sanitary landfill leachate disposal by slow rate infiltration on land which is applicable to Ontario conditions. The proposal addressed known and potential problems associated with municipal landfill leachate generation at the Muskoka Lakes site and suggested a broader assessment of possible economically and environmentally acceptable methods of leachate disposal.

3.0 Literature Review of Landfill Leachate Generation, Collection, Treatment and Disposal

3.1 Preface

As indicated in section 2.2, a primary objective of this research project was to review the literature concerning municipal sanitary landfill leachate generation, collection, treatment and disposal. Several comprehensive literature reviews have been assembled on these topics (e.g. Robinson and Maris, 1979) but some revision is necessary in light of new information stemming from more recent research activities and engineering experience.

An important aim of landfill management is to minimize leachate-related problems. Emphasis in design must thus be directed to control of leachate production at source. This may be accomplished by surface sealing, establishment of vegetation on the surface to maximize evapotranspiration of incident rainfall, diverting surface or groundwater, contour grading and containment techniques. Groundwater control techniques, as an example, are generally prohibitively expensive and hence have rarely been used to alleviate existing problems (Harrington and Maris, 1986). Consequently, the leachate treatment methods described below represent remedial solutions to sanitary landfill leachate disposal once a problem is found to exist.

The origin of municipal landfill leachate is reviewed in section 3.2. Collection of leachate from "containment" and "dilute and disperse" sites, the two major landfill types, is reviewed in section 3.3. Treatment methods subsequently examined include biological (section 3.4.2), physico-chemical (section 3.4.3) and combination biological and physico-chemical techniques (section 3.4.4).

3.2 Leachate Generation

Leachate is produced when post-infiltration rainfall becomes contaminated as it passes through the layers of waste in a landfill or when groundwater comes in contact with the refuse body (Uloth and Mavinic, 1977). The volume of leachate produced is affected by the absorptive capacity of the refuse, which is in turn a function of the degree of compaction and the nature of the wastes. From work done in the U.K., it is generally acknowledged that 1 m^3 of waste can retain about 125 ℓ of leachate before gravitational flow is initiated and a seepage problem develops (Harrington and Maris, 1986). Other important determinants of leachate volumes include local climatic conditions (i.e. rainfall, humidity, and evapotranspiration

regimes), site hydrology, local groundwater flow systems in the landfill vicinity, surface liner permeability, vegetative cover, and the landfill contour as it affects surface runoff (Boyle and Ham, 1974). In principle, water balance calculations should provide an estimate of probable leachate volume production based on the reality that any water which enters the refuse body, and does not evaporate or remain stored within it, must migrate as subsurface percolation and resurface as seepage. In practice, the leachate volume produced is difficult to accurately estimate and, for this reason, many leachate treatment plants are often designed after waste emplacement (Harrington and Maris, 1986).

3.3 Leachate Collection

In general, landfills are classified either as "containment" or as "dilute and disperse" sites. Containment landfills are located on impermeable substrata or have artificial liners which contain and store the leachate produced (U.S.E.P.A., 1980). The performance of various synthetic and natural soil cover-liners in containing landfill leachate is currently an area of active research (Schomaker, 1986). In the U.S. and some European countries (e.g. W. Germany and Italy), it is mandatory to install impermeable liners to contain leachate (Harrington and Maris, 1986).

A "dilute and disperse" landfill, such as the Muskoka Lakes site, allows leachate migration to the underlying substrata where the unsaturated zone above the groundwater table effects some attenuation of leachate contaminants (Lancashire County Council, 1984). Even though this zone is unsaturated, the conditions are anaerobic (anoxic) due to presence of landfill gases (e.g. CO_2, methane) as decomposition products (Robinson and Lucas, 1985). Further leachate treatment requires collection at a central point via subsurface tile networks or peripheral drainage. The most effective and commonly used collection/disposal system for "dilute and disperse" landfills in the U.K. is illustrated in Figure 2.1. As leachate is collected by artificial drainage, it is pumped to a holding lagoon and then distributed on adjacent land or to specially prepared grass plots where evaporation, absorption and percolation processes occur. The percolate is generally collected using subsurface or peripheral drainage and redirected to a holding tank where it can be tested for suitability for discharge to a watercourse. Unsuitable percolate is returned to the holding lagoon for re-application on land. Adoption of a feedback loop network of this type may be practicable under Ontario conditions given the increasing public

concern over water quality in the environs of the many isolated and derelict landfills in the province.

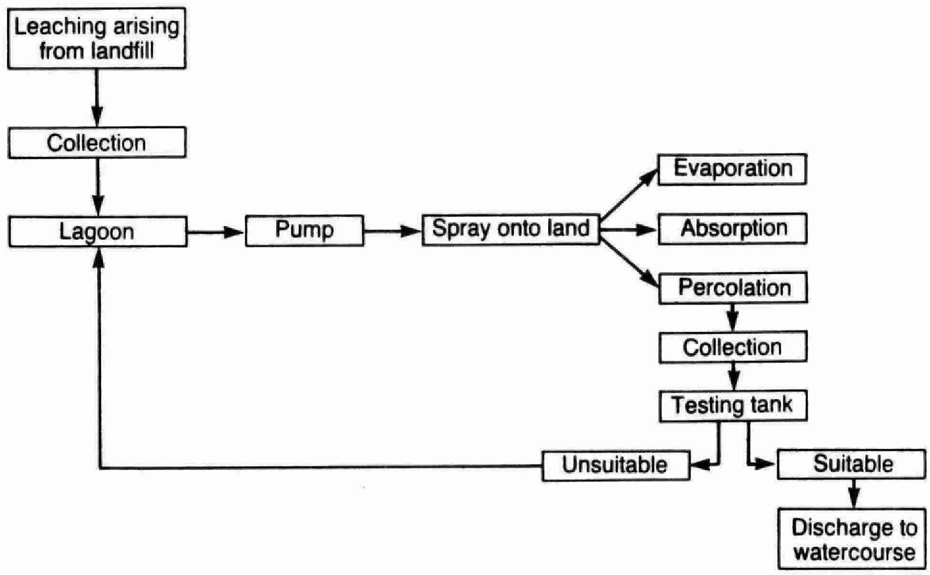


Figure 3.1. Landfill leachate collection/disposal system (after Lancashire County Council, 1984; p. 37).

3.4 Leachate Treatment and Disposal

3.4.1 Preface

Due to the composition and strength of most leachates, the most important constraint on leachate treatment is cost (Harrington and Maris, 1986). Leachate typically contains high concentrations of organic and inorganic pollutants which, along with the hydraulic load, can vary diurnally, seasonally and as the wastes age. Extremes in pH, Biological Oxygen Demand (BOD), Chemical Oxygen Demand (COD), and heavy metals often occur (Chian and Dewalle, 1976; Keenan et al., 1984).

In general, treatment systems need to be thorough, flexible and broad spectrum since the appropriate type of treatment is likely to change as the wastes age. Younger landfills tend to produce low pH leachate (i.e. a high ratio of BOD:COD). Consequently, leachate from 'young' landfills responds well to biological treatment (Tittlebaum, 1982; Menser et al., 1983;

Harrington and Maris, 1986). As the wastes age, the volatile fatty acids are converted to methane and carbon dioxide giving rise to a lower BOD:COD ratio. 'Older' leachate is more responsive to physico-chemical treatment since a high proportion of the organic material consists of relatively refractory compounds (e.g. humic and fulvic acids) (Knox, 1983; Robinson and Maris, 1985). In addition, the anaerobic reducing environment within the landfill results in high concentrations of ferrous iron (Fe^{+2}) and ammonia-nitrogen (NH_4^+) (Tittlebaum, 1982; Robinson and Maris, 1985). A combination of biological and physico-chemical processes is likely to be required to effectively treat leachate of changing quality over the longer term.

Harrington and Maris (1986) offer recommendations which are timely given the operational status of solid waste disposal technology in Ontario. At existing landfill sites, it is suggested that pilot-scale studies precede full-scale treatment installations. For new sites, leachate strength and volume should be estimated with the best available data, accounting for local precipitation regimes as well as the anticipated volume and degree of compaction of the wastes. The choice of treatment processes will then be a function of cost, effluent discharge criteria and waste age.

Sections 3.4.2 to 3.4.4 (inclusive) summarize the existing technology on biological, physico-chemical and multi-phase biological and physico-chemical treatment systems. Biological treatments include both aerobic (aerated lagoon/activated sludge) and anaerobic processes. Treatment by activated carbon, reverse osmosis, natural treatment media and coagulants-oxidants are dealt with as physico-chemical treatment options. Slow rate soil infiltration land treatment, landfill recirculation and application to wetlands are considered as combination biological and physico-chemical treatment methods.

3.4.2 Biological Treatment

3.4.2.1 Aerobic Treatment

Aerobic types of treatment effect biological transformation of hazardous compounds to more innocuous substances. Mineralization is the primary mechanism whereby organics are converted to carbon dioxide and water. The simplest aerobic treatment consists of a diffused aerator within an aeration tank/lagoon.

Activated sludge culture units employ the suspended growth of microorganisms to flocculate fine colloidal solids present in the incoming wastewater (Keenan _et al._, 1984). The treated and untreated wastewaters are

then separated by gravity sedimentation in a clarifier. In addition, nitrifying organisms can produce an effluent with very low concentrations of ammonia. Activated sludge cultures are limited by phosphorus availability and inhibited by ammonia toxicity. Before exposing the leachate to activated sludge, Keenan et al. (1984) recommend i) lime pretreatment for iron removal, ii) phosphoric acid addition to neutralize the lime and provide P for the activated sludge, and iii) ammonia removal by air-stripping.

Early studies of simple aerobic treatment of leachate gave encouraging results. After a 10-day retention period within an aerated beaker, 90.0 - 99.7% and 97.0% of the BOD and COD, respectively, were removed (Boyle and Ham, 1974; Cook and Foree, 1974). Since the removal of organic matter, ammonia and metals occurs largely within a settled biological floc, sludge control would be required at a treatment plant scale (Uloth and Mavinic, 1977; Robinson and Maris, 1985).

For effective aerobic treatment of leachate, several conditions must be controlled. Leachate strength and flow variations should be reduced by "balancing". This simply involves collecting leachate in a holding lagoon before it is treated (Uloth and Mavinic, 1977; Harrington and Maris, 1986). The maintenance of aerobic conditions and the addition of P to low phosphorus leachate would also be required for adequate treatment (Robinson and Maris, 1985). Decreased hydraulic retention time might be possible for a relatively stable and low BOD leachate.

Aerated lagoons are generally preferred to the activated sludge process due to simplicity and cost (Harrington and Maris, 1986). Since biological filters accumulate iron and organic growth, they have not been entirely successful in treating 'young' leachates. The use of rotating biological contactors for nitrification of ammonia in 'older' leachates may be a viable aerobic treatment option in the future (Harrington and Maris, 1986).

3.4.2.2 Anaerobic Treatment

Methanogenesis is the principle process involved in anaerobic treatment to convert organic contaminants to carbon dioxide and methane. Boyle and Ham (1974) reported greater than 90% BOD removal from landfill leachate when the hydraulic retention time was greater than 10 days and the temperature was maintained between 23-30°C. Temperature greatly affects stabilization in the lower range of 11-23°C. Harrington and Maris (1986) have indicated that quiescent lagooning greatly reduces organic loading but potential odour

nuisances and the large area required are disadvantages to this type of treatment. Development of upward anaerobic sludge-blanket reactor (UASB) technology may prove to be suitable for high strength wastewater treatment (Kennedy and Guiot, 1986).

3.4.3 Physico-Chemical Treatment

Physico-chemical treatment is more effective than biological treatment when the BOD:COD ratio is low as is typical of 'older' leachates (Chian and Dewalle, 1976; Knox, 1983). Since chemical treatments are highly specific, the characteristic variability of leachate quality makes certain of these forms of treatment impractical.

3.4.3.1 Activated Carbon

The extremely high internal surface area and intricate pore structure of activated carbon (bituminous coal) allows it to both filter suspended solids and adsorb dissolved organic material of a broad range of molecular weights onto its surface. This treatment process can efficiently remove many priority pollutants and hazardous substances such as the BTX compounds, phenols, dieldrin, polychlorinated biphenols (PCB) and refractory organic acids (Dewalle and Chian, 1974; Ho et al., 1974) Activated carbon has achieved COD removal efficiencies of 34-59% for untreated leachate (Cook and Foree, 1974; Ho et al., 1974; Harrington and Maris, 1986) and 81% for lime pretreated leachate (Uloth and Mavinic, 1977). Colour and iron removal are also achieved with such high surface area substances but renovation of high strength wastewaters is expensive due to re-activation and replacement costs (Knox, 1983). Frequently, activated carbon is used as a final polishing stage subsequent to treatment by reverse osmosis.

3.4.3.2 Reverse Osmosis

Reverse osmosis (RO) has been used successfully for many years to desalinize brackish waters and brine (Applegate, 1984; Cartwright, 1985). More recently, RO technology has been applied in the development of portable units capable of decontaminating sites where localized chemical spills have occurred. RO requires that extreme pressure be applied to the solution on one side of a semi-permeable membrane. The spontaneous osmotic flow is thus reversed giving rise to a concentrated 'permeate' of approximately 1/10th of the original wastewater volume, the remainder being a highly treated effluent. As mentioned above, this process is often coupled with activated

carbon filtration as a final polishing step. Removal of 90% of the organic solutes can be obtained if their molecular weight is in excess of 120. At molecular weights of less than 120, separation of organics from the solution decreases significantly (Applegate, 1984).

RO membranes and devices are susceptible to biological slime and scale formation, fouling by suspended solids and colloidal materials as well as oil and metal oxide deposition (Rak, 1984). The likelihood of RO membrane failure is high if pretreatment by ultrafiltration is ignored (Rak, 1984; Shaw, 1985). The feed stream must be compatible with the membrane and other components in the system.

The chemical compatibility of RO membranes is known for most substances found in seawater but the membrane type and configuration suited to removal of various priority pollutants is only now being investigated (see section 8.2.2). Little is known about the suitability of RO technology for landfill leachate treatment. It is expected that improved membranes will decrease the present RO application limitations. RO will undoubtedly represent an important advancement in the area of water reuse and pollution control in the future (Applegate, 1984). The temporal variability of leachate strength, however, may pose a technological constraint as it has to other forms of physico-chemical treatment.

3.4.3.3 Natural Treatment Media

Numerous natural organic substances such as peat moss have the capability of chelating and immobilizing metals from solution. Tree bark treatment offers an inexpensive and irreversible method of removing ferrous iron from certain types of wastewaters (Vaughan et al., 1984). Up to 65 mg of elemental iron can be absorbed per gram of dry bark. Since only small quantities of soluble phenolics are present in weathered conifer bark in particular, the treated effluent is usually sufficiently free of these substances to allay any environmental concerns (see section 8.2.2).

3.4.3.4 Coagulants - Oxidants

A wide variety of chemicals have been identified which can precipitate, coagulate, oxidize or reduce organic and inorganic constituents in landfill leachate. In isolation, such treatments have met with limited success. A series of chemical treatments have been shown to be more effective but require a high degree of control over the reaction conditions.

Acceptable iron and colour removal can be achieved with precipitation by lime and sodium sulfide and coagulation by alum and ferric chloride at high application rates. High degrees of iron removal (i.e. post-treatment concentrations of 20 mg $Fe \cdot \ell^{-1}$ from initial levels of 300 $mg \cdot \ell^{-1}$) have been demonstrated with lime additions of 840 $mg \cdot \ell^{-1}$. The corresponding increased sludge production, however, is an undesirable by-product. The removal of organic matter, chloride and particulates by these agents have been relatively ineffective at less than 30% reduction (Ho et al., 1974; Chian and Dewalle, 1976).

In a laboratory study by Ho et al. (1974), oxidation by calcium hypochlorite, chlorine and potassium permanganate effected COD removal of 48, 23 and 20%, respectively. At the same time, iron and colour removal were excellent. Undesirable aspects of such treatments include increased total solids, hardness and chloride concentrations.

Hydrogen sulphide, the unpleasant odour frequently associated with open bodies of leachate, can be effectively removed with hydrogen peroxide (Fraser and Sims, 1983). This method is used extensively in the U.K. to control leachate odour.

3.4.3.5 Physico-Chemical Treatment Summary

It is widely recognized that physico-chemical treatments have been more effective in the controlled setting of the laboratory than in field applications. The high degree of control required for such treatments, the variability of leachate strength and the sludge or reaction by-products make many physico-chemical treatments somewhat unattractive. While no single physico-chemical treatment lends itself to full leachate treatment at this time, the advancement of activated carbon and reverse osmosis technologies appears extremely promising.

3.4.4 Combination Biological and Physico-Chemical Treatment

Keenan et al. (1984) point out the principal reason for the poor removal efficiency of biological forms of treatment in the absence of physico-chemical pre-treatment. Biological treatment is necessary to stabilize the degradable organics but is inhibited by the presence of metals. Physico-chemical treatment is needed to remove these metals and to hydrolyze the less degradable organics. Thus, combination treatments have produced the most encouraging treatment results.

Following a complex multi-phase treatment, reductions of BOD, COD and ammonia as high as 99, 95 and 90%, respectively, were reported by Keenan et al. (1984). This treatment involved metal cation precipitation by lime, nitrification of excessive ammonia by air-stripping and neutralization with sulphuric phosphate. Once the heavy metals and excess ammonia were removed, microorganisms of an activated sludge secondary treatment reduced BOD concentrations by 99% to 153 mg BOD.ℓ^{-1}, as long as the process occurred above 10^{0}C. In another case, biological treatment was followed by liming (Ho et al., 1974). The result was 97% COD removal and virtual elimination of iron and colour.

Three other combination treatment/disposal methods include slow rate infiltration land treatment, recirculation of leachate to the landfill and leachate disposal in wetlands. Forms of slow rate infiltration land treatment include trickle, subsurface, and spray irrigation. The latter is the most commonly used form of land treatment in the U.S. (Nordstedt et al., 1975; Menser, 1981; Prouty, 1986) and the U.K. (Harrington and Maris, 1986). Spray irrigation is the treatment aimed to maximize organic contaminant volatilization and water evaporation. Trickle irrigation provides good surface distribution and optimizes use of the most microbially-active soil zone (i.e. high organic matter topsoil or litter layer) in leachate renovation while eliminating direct phytotoxic foliar contact with vegetation (Oron et al., 1986). Subirrigation is the most aesthetically acceptable application technique in that it eliminates odour emissions and spray drift which can create a "nuisance" to local residents (Schauer, 1986). Distribution through a subsurface tile network also affords the possibility of winter applications depending on local frost depth in the soil (Harris, 1978). This technique, however, does circumvent leachate rejuvenation in the microbially-active topsoil and lower subsoil temperatures may curb microbial activity despite bioactive leachate introduction.

Recirculation of leachate through the landfill has also been proven effective in the U.S. (Pohland, 1980; Tittlebaum, 1982) and in the U.K. (Newton, 1979; Rowe, 1979; Barber and Maris, 1984). Wetland ecosystems have shown potential in providing municipal and industrial wastewater treatment (Tilton and Kadlec, 1979; Hantzsche, 1985; Richardson and Nichols, 1985; Wile et al., 1985; Gersberg et al., 1986). Natural or artificial wetlands may thus offer a treatment option for landfill leachates.

3.4.4.1 Slow Rate Infiltration Land Treatment - Spray Irrigation

Land treatment systems can be categorized into three main types according to the mode and rate of wastewater application; slow rate infiltration, rapid rate infiltration and overland flow (Iskander, 1981; C.R.R.E.L, 1984). Each is characterized by a unique hydraulic loading rate and set of ecosystem component requirements to achieve optimum wastewater treatment. The slow rate infiltration method makes the best use of intrinsic ecosystem attenuation capacity by:

(i) maintaining an aerobic soil environment, maximizing evapotranspirational losses and minimizing deep percolation (i.e. unlike the rapid rate infiltration method), and

(ii) by eliminating the need in most cases for recollection and possibly final polishing treatment before discharge into watercourses (i.e. unlike the overland flow method).

A thorough review of the three methods has been conducted by Zirschky et al. (1986) and is reproduced in Appendix A.

Spray irrigation, one form of the slow rate infiltration technique, offers several mechanisms for reducing the pollution hazard of leachate applied to land. Atmospheric evaporation is maximized (i.e. 400-500 $mm \cdot yr^{-1}$; Lancashire County Council, 1984) by spraying for short periods (i.e. 5-15 minutes per spray cycle; Rowe, 1979). Transpiration by vegetation further reduces leachate volume. At the same time, the addition of nutrients from leachate to the plant root zone can improve the productive capacity of marginal lands (Menser et al., 1983; Lancashire County Council, 1984). The physical processes of soil infiltration and interflow disperse and dilute the leachate in the unsaturated zone. Furthermore, inorganic compounds are chemically precipitated or adsorbed on soil colloid exchange sites and the organic contaminants are decomposed by the soil microbial population. In short, the soil/microbial/vegetation ecological system acts as a "living filter".

A considerable body of information exists on spray irrigation of municipal sewage wastewaters in both grassed and forest ecosystems, and has been recently reviewed (Anonymous, 1985). However, disposal of landfill leachate by irrigation is a much more recent practice and the effects on vegetation and soils are not well documented. Like sewage wastewaters,

landfill leachate contains all essential plant nutrients, albeit in lesser quantities, and would seem to have good potential for recycling through biophysical systems. Phytotoxic effects have been observed but it is not yet clear whether this is a function of waterlogging, leachate constituents or other factors.

Early work by Menser (1981) indicated that direct contact of leachate with tree foliage was phytotoxic. Although the physiological mechanisms were unknown, it was believed that accumulation of iron, manganese and heavy metals were largely responsible. For this reason, spray irrigation of leachate in the southern U.S. is carried out during the vegetatively dormant months (October to April) when direct foliar contact can be avoided. Later work by Menser et al. (1983) suggested that leachate was not an important source of metal contamination in six forest species studied. While total N, P, K and Cl were consistently higher in all tree species after irrigation, Fe was largely unavailable to plants. Manganese and Al absorption also decreased with leachate applications. Increased soil pH could explain low or decreased availability of these metals to plants. Iron accumulation on the leaf surface has been suggested as a cause of plant mortality by Menser et al. (1983). This view is not shared by certain U.K. research groups (Rowe, 1979; Harrington and Maris, 1986) where vegetative staining is thought of simply as aesthetically unsightly. There are clearly a number of crucial areas for which no concensus has been reached in the literature.

Problems identified by other researchers, mainly in the U.S., may have significant implications for the prospects of spray irrigation operations in Canada. These aspects include i) application problems in sub-zero temperatures, ii) increased likelihood of soil waterlogging due to lesser potential evapotranspiration rates, iii) creation of a subsurface placic layer in coarse-textured, acid soils, iv) highly variable soil attenuation capacities, and v) variable sensitivities of indigenous vegetation types to leachate constituents.

Spraying cannot be carried out when the air temperature drops below 0°C. Therefore, the technique of spraying when vegetation is dormant (October to April) is not feasible in this country. Furthermore, low temperatures tend to reduce the degree of renovation achieved in the soil (Harris, 1978). In a moderate climate, the rate of renovation can be improved by adding activated sludge to the leachate prior to irrigation (Lancashire County Council, 1984).

It is essential that application rates be flexible and controlled to vary in accordance with ambient weather conditions and with longer term vegetation and soil changes. Overspraying leading to waterlogged soil conditions is undesirable since plant transpiration can actually be inhibited by root disfunction in a poorly aerated soil environment. This will lead ultimately to other adverse plant physiological responses affecting growth.

Prolonged spraying on granitic, coarse-textured soils undergoing natural podzolization with iron-laden leachate tends to cause the formation of an indurated layer in the B horizon which impedes water movement and root growth (Lancashire County Council, 1984). This effect has been widely observed at the Muskoka Lakes site and is discussed in section 6.4.4. In intensely managed grassed ecosystems, this problem can be overcome to some extent by regular subsoiling operations.

The litter layer of a forest soil is important to pollutant attenuation. Soil analyses have shown that most of the leachate-supplied nutrients are retained in the surface 5 cm of soil due to the mechanisms of dispersion, dilution and cation exchange (Bennett et al., 1975). The litter layer moderates the rate of soil infiltration while reducing the erosion hazard presented by soils artificially maintained in a wet condition (Rowe, 1979).

Finally, Knox (1983) suggests that chloride, total dissolved solids and electrical conductivity may affect achievable evaporation rates and may indicate that salt-tolerant species should be selected for cultured spray areas. A variety of ecosystems have shown encouraging results for leachate disposal by spray irrigation. These include unimproved scrublands (Lancashire County Council, 1984) and grasslands (Menser et al., 1983; Nordstedt et al., 1975). There is a very real shortfall evident in the literature, however, in research related to comparative vegetation sensitivities to wastewater disposal.

3.4.4.2 Recirculation of Landfill Leachate

Recirculation of landfill leachate through the refuse body has been recognized as a method of providing leachate treatment while controlling surface and groundwater pollution associated with sanitary landfills (Pohland, 1980; Tittlebaum, 1982; Robinson and Maris, 1985). It is acknowledged that recycling provides a desirable moisture content in the decomposing wastes and allows for continual exposure of biological populations within the landfill to leachate nutrients. This results in

accelerated anaerobic decomposition and landfill stabilization which are evidenced by a rapid decline in the strength of the leachate produced (Robinson and Maris, 1985). Pilot-scale studies have shown that 97% of COD can be removed by recirculation, while on site studies have indicated a more modest 40% reduction in COD over a 20-month period. As COD decreased, reduced concentrations of volatile acids, Fe, Mn and the heavy metals were associated with a rise in pH (Robinson and Maris, 1985).

Recirculation allows the 2-step process of acid fermentation to take place within the refuse. Organic pollutants are transformed to volatile acids which in turn are converted to methane and carbon dioxide (Pohland, 1980). Nutrient and trace elements can either be assimilated biologically or attenuated by physico-chemical interactions. Overall, residual contaminant concentrations in the stabilized leachate can be reduced and removed via precipitation, sorption, filtration and chemical "complexation" within the refuse body (Pohland, 1980).

The collection and recycling of leachate is an advantageous landfill management option because it promotes rapid, predictable stabilization of readily degradable organic wastes as evidenced by an increase in the rate of landfill gas production. Furthermore, recirculation of the leachate to the landfill surface by spray irrigation can result in a significant decrease in leachate volume due to evapotranspirative losses (Lee et al., 1986). Recirculation is also beneficial in that rapid landfill stabilization decreases the amount of time required before the landfill site can be converted to an alternative land use (Maye, 1972).

3.4.4.3 Leachate Disposal on Wetlands

Many of the natural functions performed by wetlands in an ecosystem environment (i.e. water retention and purification) are consistent with the requirements of wastewater treatment. Mineral soils and organic detritus physically filter pollutants. Microorganisms utilize and transform contaminants to more benign forms, while aquatic and riparian plant species serve to assimilate the available nutrients produced (Hantzsche, 1985). Nitrate assimilation by both plants and microbes, denitrification and phosphorus removal by biological means and physical sorption are some of the major processes involved (Tilton and Kadlec, 1979).

Experience has shown that disposal of wastewater in natural or artificial wetlands has been most successful when some form of pretreatment was carried out (Hantzsche, 1985; Wile et al., 1985). This frequently

involves coarse screening followed by a 5-10 day retention period in an alum-fed aerated cell. This decreases BOD, suspended solids and P and prevents sludge accumulation in the wetland. Other considerations for success with this method include high length-to-width wetland configuration, shallow surface water and hydraulic loading rates no greater than 200 $m^3 \cdot ha^{-1} \cdot d^{-1}$ (Wile et al., 1985). More research is required to determine the plant species which are most effective at nutrient removal and least sensitive to the more persistent residual substances in wastewater (Wile et al., 1985; Gersberg et al., 1986).

3.5 Conclusions

In general, leachate treatment can be viewed as a necessary remedial measure in the overall management of sanitary landfills, both before and after decommissioning. No single biological or physico-chemical treatment has yet emerged that will deal effectively with the temporal variability of leachate strength and volume. Several combination techniques appear promising with the best alternative being biological pretreatment (aerated lagoon) of 'young' leachate or physico-chemical pretreatment (activated carbon or reverse osmosis) of 'older' leachate followed by slow rate infiltration land treatment. Recirculation of leachate to the landfill appears feasible as a solution to the winter season disposal constraints in Ontario. Should slow rate land infiltration be adopted as a major management option for many of Ontario's landfills in the future, then the feedback loop of testing percolate quality after land infiltration should be recognized as an important extension of this overall treatment scheme. A great deal must yet be learned from experience elsewhere and from research specific to Ontario biophysical and climatic conditions before such a commitment can be made or suitable design guidelines established.

4.0 General Site Description

The Muskoka Lakes sanitary landfill site (known locally as the Pinelands site) is situated on Lots 21, 22 and 23, Concession 1 of the Medora Ward in the Township of Muskoka Lakes, District Municipality of Muskoka. The property is approximately 400 metres from the North Bay of Lake Muskoka and is accessible from Muskoka Rd. 26 (Figure 4.1).

For the most part, the topography is irregular and conforms to the general configuration of the underlying bedrock (Figure 4.1). The local parent rock has been classified as migmatite which consists of Precambrian age amphibolitic material with granitoid bands (Dr. W. Chesworth, pers. comm.) Locally, this bedrock type is not believed to be inordinately prone to jointing as evidenced by close examination of many of the bedrock exposures in the area. The contrasting weathering characteristics of these two interbanded igneous materials, however, will undoubtedly cause fracturing regionally. Secondary permeability along bedrock fracture lines is thought to be of minor importance in the vicinity of the landfill.

The overburden is variable in depth ranging from 0 m. at frequent outcrops, through thin veneers to quite deep deltaic sand deposits (Figure 4.2). The soil is best described as relatively homogeneous medium to fine sands with occasional textural discontinuities of coarse sand and gravel. The reported permeability of the indigenous soil ranges from 6.5×10^{-4} to 1.0×10^{-2} $cm \cdot s^{-1}$ resulting in groundwater migration of 30-60 metres annually (Gartner Lee Assoc. Ltd., 1977).

Groundwater flow is controlled by the bedrock physiography and is thus local rather than regional in nature. A natural ravine situated immediately to the north of the landfill area acts as a discharge area for much of the groundwater flow originating from the north, thus diverting it into a perennial stream and on to the North Bay of Lake Muskoka. To the east of the landfill, a bedrock ridge diverts groundwater flow around the landfill, thereby further securing the natural isolation of the refuse body from extensive contact with local groundwater flow systems.

The forest cover is mixed hardwood made up predominantly of an uneven-aged hard maple/beech stand (i.e. shade tolerant species) with some aspen and hemlock. The understory is typical of an upland hardwood forest in the region. A detailed account of the diversity of plant species making up the ground cover on site can be found in section 7.2.2.

Dr. W. Chesworth, Professor of Geology, Department of Land Resource Science, University of Guelph, Guelph, Ontario

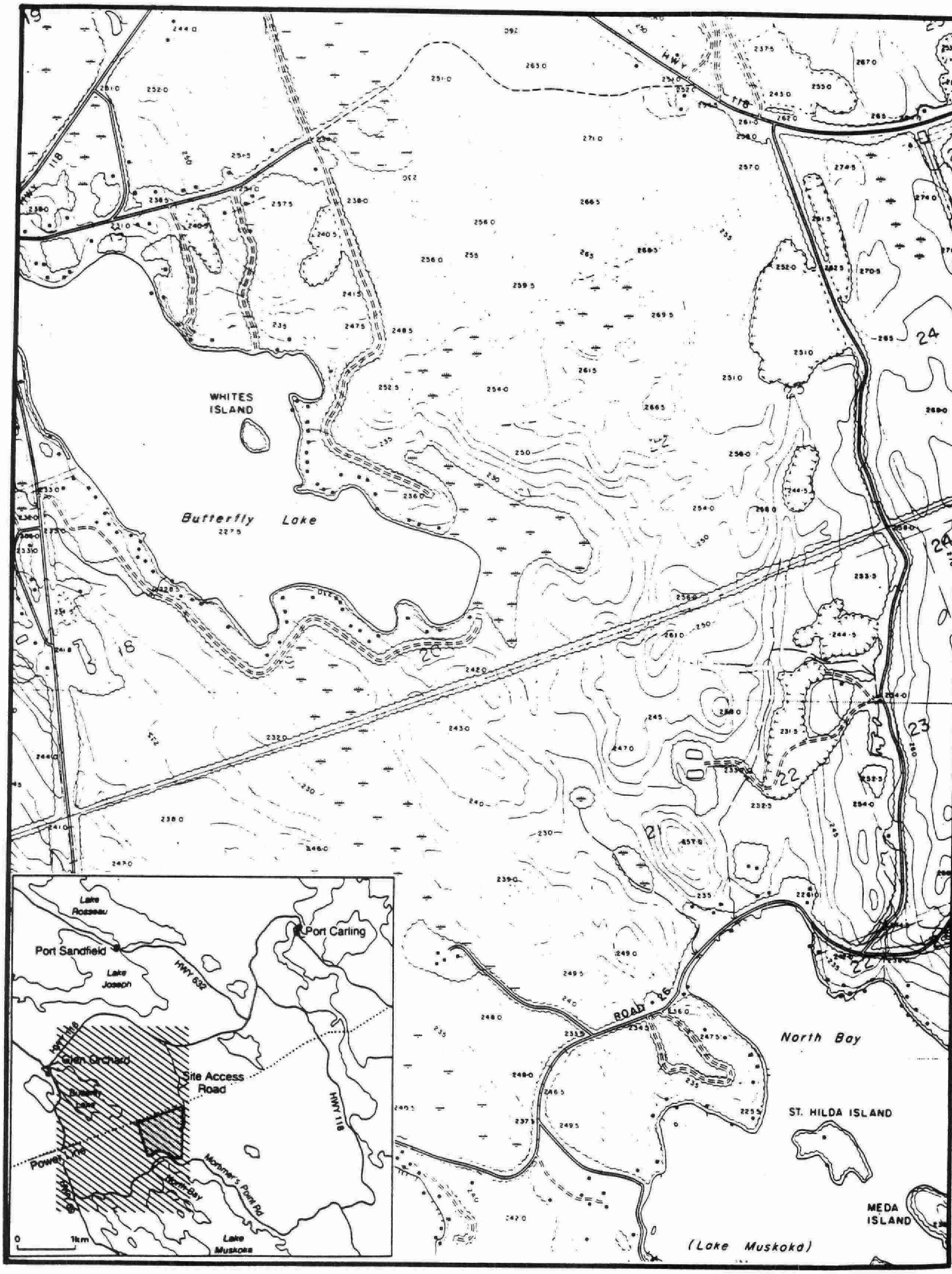
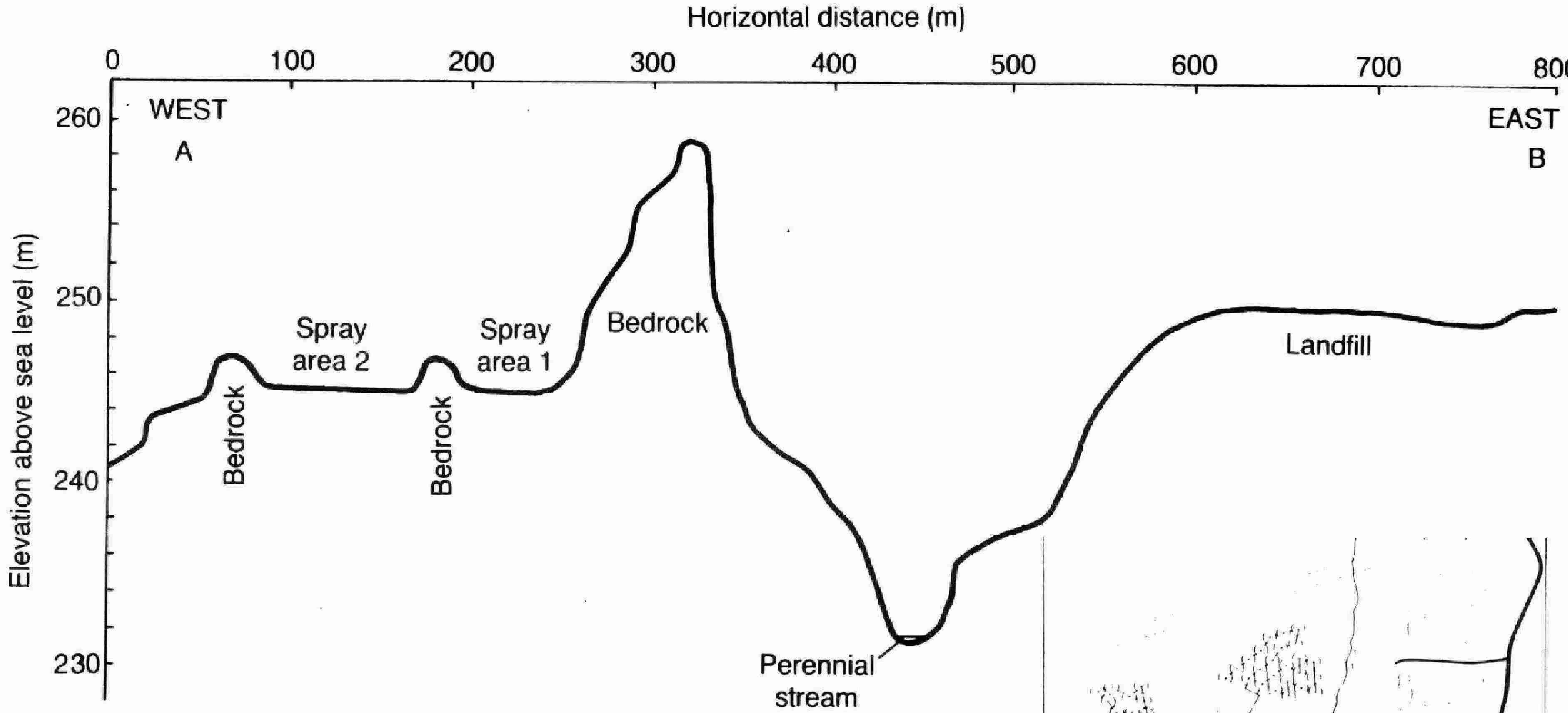


Figure 4.1. Section of the Ontario Base Map (O.B.M.) for the landfill vicinity based on 1984 aerial photography (reduced from 1:10,000 published scale to 1:16,600).



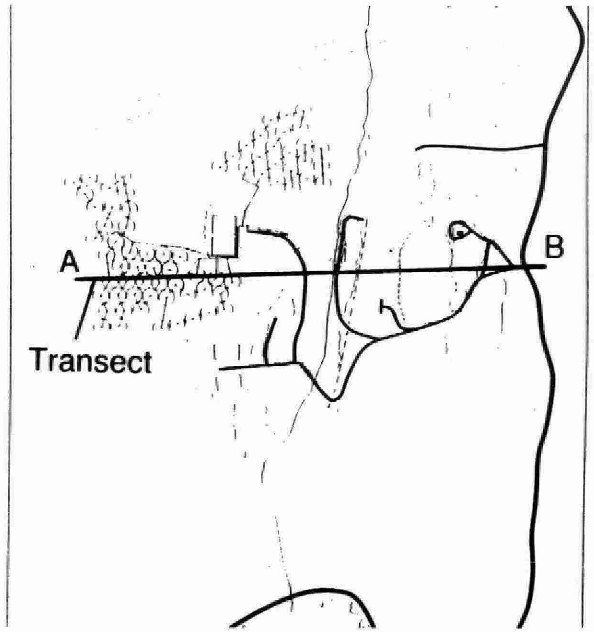


Figure 4.2. Elevational cross-section along an east-west transect through the municipally-owned lands at Muskoka Lakes.

The total annual precipitation in 1986 as recorded by the Muskoka Airport A.E.S. meteorological station near Bracebridge was 1158.1 mm. This is about 17% higher than the 30-year climatic norm of 993.1 mm for this area. During the six-month field season from May to October in 1986 (Figure 4.3), recorded monthly precipitation totals were 102.8 mm, 127.2 mm, 103.7 mm, 96.0 mm, 174.7 mm and 103.5 mm, respectively. In order, these values represent increases of 30%, 64%, 20%, 27%, 75% and 13% when contrasted to 30-year precipitation normals for these months. In light of these high monthly precipitation totals, it is evident that 1986 was an abnormally wet field (i.e. spray) season at the Muskoka Lakes landfill.

Figure 4.4 shows the relationship between the long-term local precipitation patterns recorded at Muskoka Airport and simple temperature-based (Thornthwaite) estimates of potential evapotranspiration (E_p) on a monthly basis. Assuming a soil moisture storage capacity of 100 mm of water, Figure 4.4 would suggest that one would not expect a seasonal moisture deficit to occur on an average year. Monthly E_p amounts exceed precipitation only in the peak summer months of June, July and August and at no time is the soil moisture storage depleted sufficiently to suggest that a moisture deficit exists. Soil moisture reserves are recharged primarily in September and October with the balance of the year in a surplus soil moisture condition. A surplus indicates that the soil is at "field capacity" and excess precipitation is lost as deep drainage and/or surface runoff.

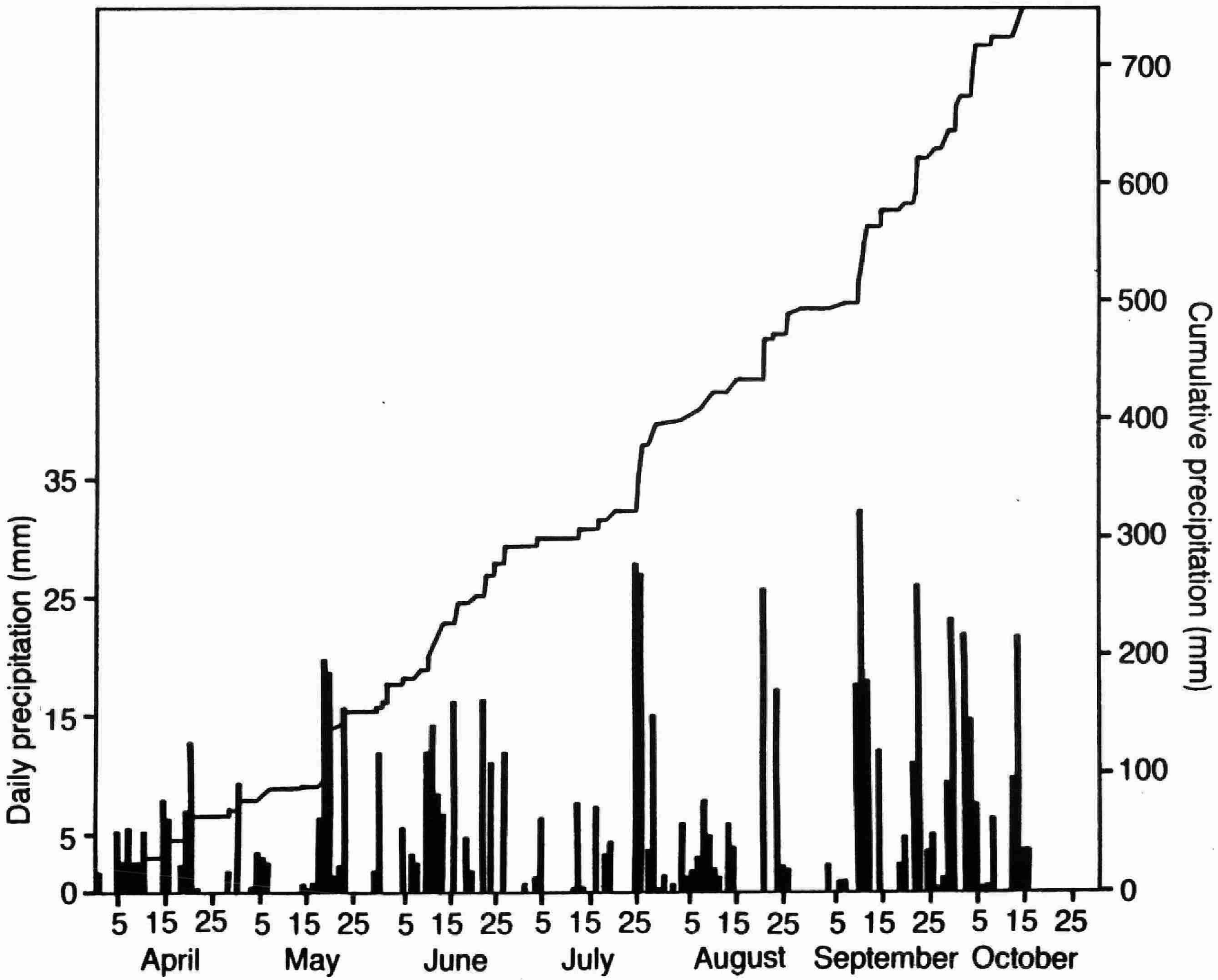


Figure 4.3. Daily and cumulative precipitation recorded at Muskoka Airport in 1986.

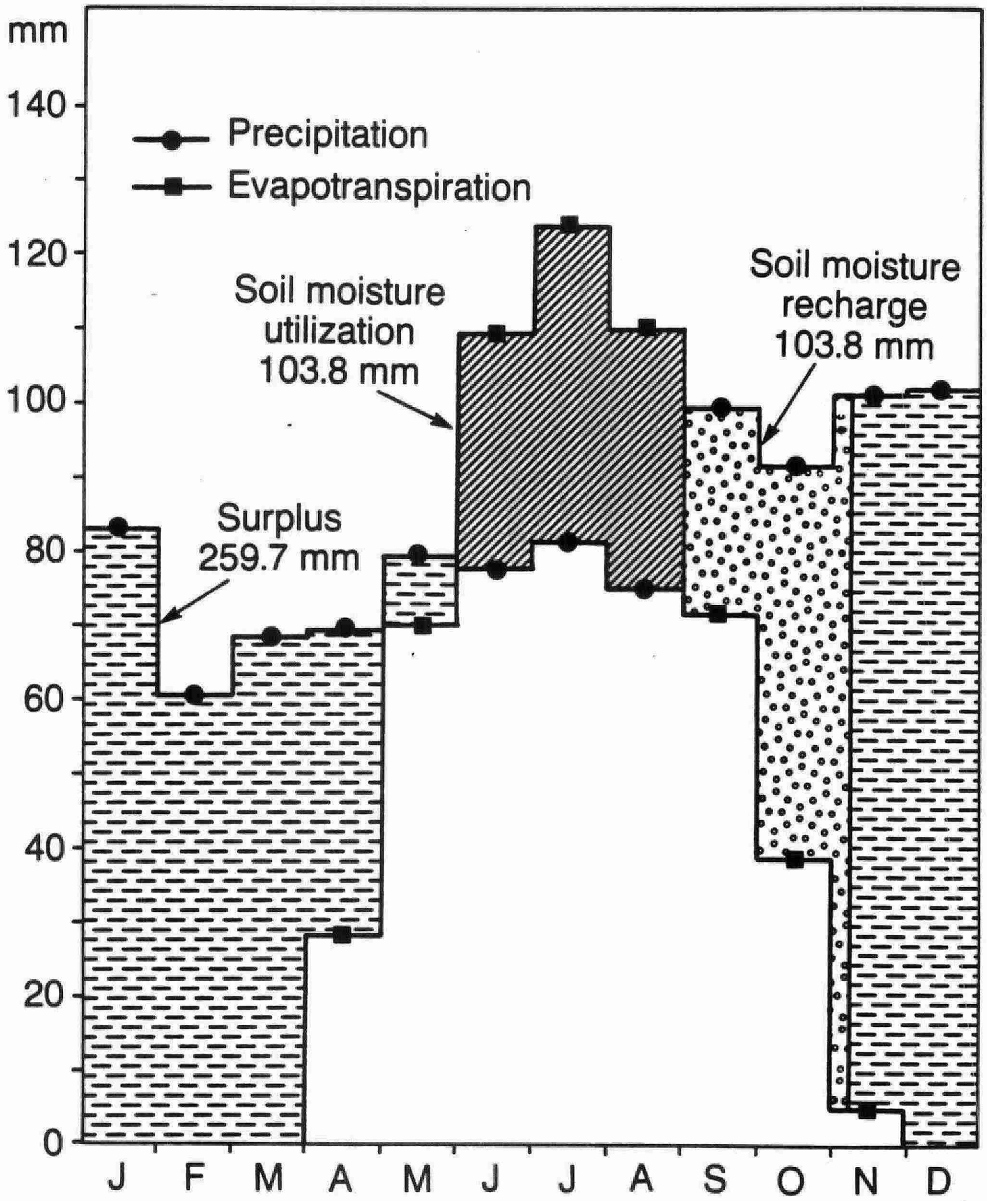


Figure 4.4. Monthly moisture budget based on 30-year climatic normals (1951-1981) from Muskoka Airport.

5.0 Study of Site Hydrology and Leachate Volume Reduction

5.1 Preface

It has been estimated in earlier hydrological reports that, at current rates, 2.4×10^7 ℓ or more of leachate will be generated and collected annually at this landfill for years to come. Given the maximum current disposal capacity of approximately 2.0×10^7 ℓ under ideal conditions (i.e. spraying not abated due to rainfall) between April and October, the existing disposal facility is clearly inadequate over the long-term (Gartner Lee Assoc. Ltd., 1984). Only winter season applications could bring these figures in line and reduce the large surplus caused in practice by spraying being interrupted by rainfall. Further, given the necessary and rapid expansion of the actual spray area in recent years and the uncertainty regarding the long-term attenuation capacity of the shallow sandy soils available for land disposal, it was apparent that a significant research effort was required to identify means of reducing the total volume of leachate generated.

It was known from previous hydrogeological investigations of the site before landfilling began that this location provided only a limited margin of protection against groundwater contamination. This is due to the high permeability of the deltaic sands on site in conjunction with shallow groundwater table levels and an approximate hydraulic potential gradient (dh/dl) of 0.12 in the direction of the perennial stream (M.O.E., 1975).

More detailed investigations were undertaken in 1986 to determine the permeability of the landfill cap in terms of its effectiveness in curbing leachate generation arising from infiltration of incident precipitation (section 5.2). Saturated hydraulic conductivities of the contaminated aquifer were determined using a variation of the auger hole method. Groundwater flow directions and velocities were also investigated as part of an overall hydrogeologic evaluation of the site (section 5.3). The feasibility of lowering the groundwater table beneath the landfill area was examined as a final step in achieving hydrologic isolation of the refuse body.

It should be noted, however, that this section of the report does not constitute a full hydrogeological investigation of the site. Emphasis in this study was placed on the use of physical and chemical properties in the unsaturated vadose zone as indicators of existing or potential groundwater contamination.

5.2 Reduction of Net Infiltration Through the Landfill Surface Liner

5.2.1 Condition of the Existing Landfill Surface Liner

5.2.1.1 Clay Distribution and Depth

The permeability of the landfill cap was investigated to determine its effectiveness in eliminating leachate generated by precipitation. The landfill surface was systematically probed (i.e. grid survey) to determine the depth of the clay cap which was generally encountered within 10-15 cm of the grassed surface. Approximately 800 manual probe inspections were made and the depth (occurrence) data were recorded.

As can be seen in the Figure 5.1 transparent overlay, not only is there considerable variance in the depth of clay where present, but the layer is also discontinuous with 19% of the area having no clay cover whatsoever (Figure 5.2). A lack of locally available clay suitable for a capping material was in part responsible for the poor integrity of the surface liner. Another observation was that at a large number of locations sand was mixed with the clay, which would cause the permeability to be higher than desirable.

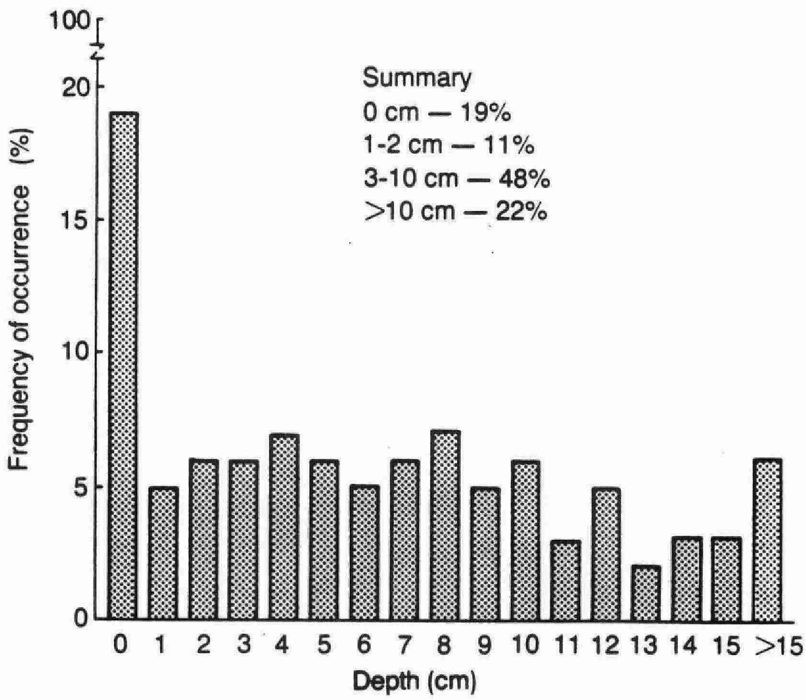
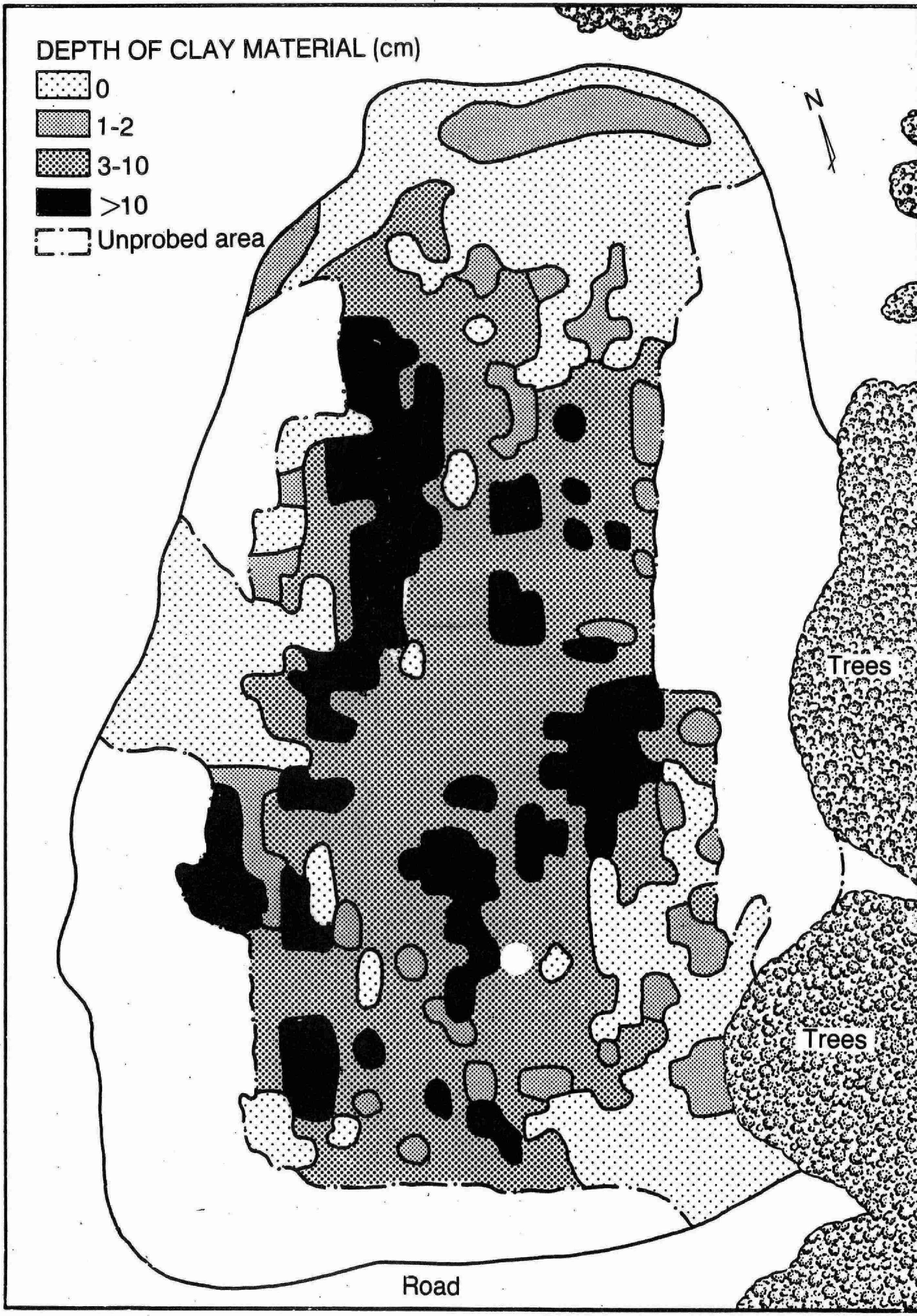


Figure 5.2. Frequency distribution of clay depth on the landfill.



DEPTH OF CLAY MATERIAL (cm)
0
1-2
3-10
>10
Unprobed area
N
Trees
Trees
Road

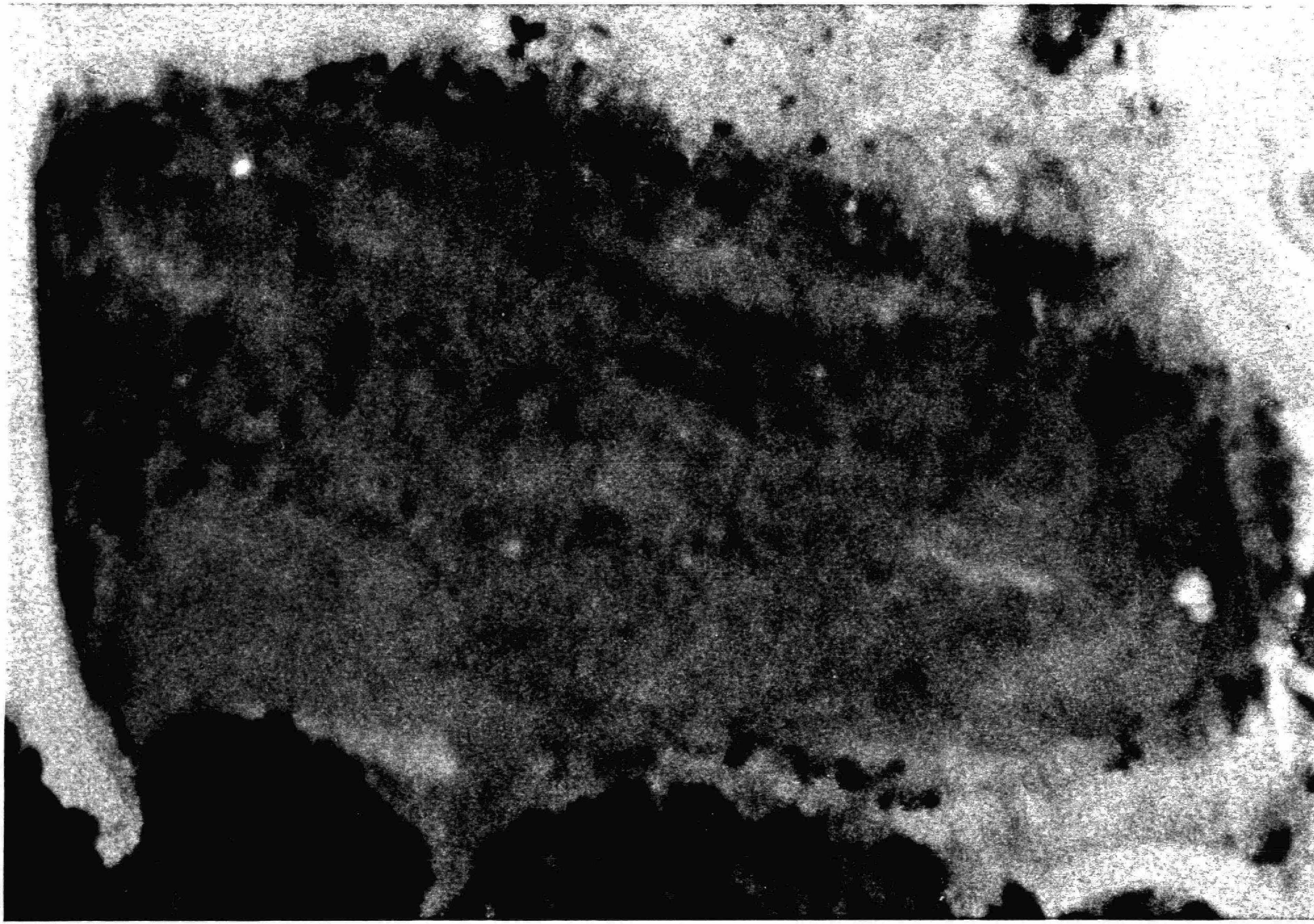


Figure 5.1. Survey of existing landfill cap condition from intensive soil sampling (transparent overlay) and false-colour infra-red aerial photography taken in June, 1986 (photographic image).

Low altitude false-colour infra-red photography strongly supported the manual cap probing results (Figure 5.1). The reflected IR-sensitive images identified areas of differential vegetative moisture stress on the grassed landfill surface. Clay-capped areas retained higher quantities of plant-available soil moisture in reserve for summer stress periods (i.e. "dry spells") whereas areas without clay coincided with areas of visible vegetative moisture stress. Actively metabolizing and transpirating vegetation generally appears magenta on such photography while plants subject to deficit water stress can be identified by a shift in hue to green or blue-green (Murtha, 1972). Much of the eastern sector of the landfill area clearly shows this change in hue which is in stark contrast to much of the western sector of the cap, and particularly to the wooded area at the perimeter of the photograph (Figure 5.1). Further evidence of vegetative stress symptoms was contributed by time-sequencing this photography. The IR images taken during four monthly flights from May to August clearly demonstrated increasing contrast in the degree of vegetative stress on the landfill cap with time through the drier summer months.

5.2.1.2 Hydraulic Conductivity of the Surface Liner

Saturated hydraulic conductivities (K_{sat}) obtained on intact cores using the constant head method in the laboratory confirmed both the impermeability of the clay material if properly applied to an adequate depth (minimum measured $K_{sat} = 5.1 \times 10^{-8}$ cm•s^{-1}) and its unsuitability for this purpose if mixed with local sandy soils in thin lifts (maximum measured $K_{sat} = 5.5 \times 10^{-4}$ cm•s^{-1}). Design specifications for clay properties and depth are most frequently cited for bottom liners (U.S.E.P.A., 1980). Assuming adequate erosion control, however, a surface clay liner of 0.5 to 1.0 m would generally be considered suitable under Ontario's climatic conditions. This section thus investigates the existing clay cap properties while recognizing the shortfall relative to minimum depth criteria.

Early in the field season (June), eight intact soil cores (5.0 cm i.d. x 3.0 cm high) were sampled from the clay cap at two randomly selected locations in the southern sector of the landfill area. Two replicate cores were taken in horizontal and vertical orientations at each location to

ascertain whether or not the mechanically emplaced clay cap was isotropic with respect to its hydraulic transmissibility characteristics. Table 5.1 shows that the K_{sat} values exhibit little anisotropic behaviour, although the horizontally sampled cores at location B possessed more variable conductivities. Closer inspection of these two cores revealed the presence of a continuous thin sand lens through the core with the more rapid K_{sat} value (i.e. 1.5×10^{-4} $cm \cdot s^{-1}$). It would be expected that the horizontally sampled cores would possess more rapid K_{sat} values due to the likelihood of some sand intermixing with the capping material in the thin horizontal lifts. Hand-texturing also corroborated the higher sand content in the location B samples, since the clay loam textural class can have up to 45% sand content. The consistency of the dry bulk densities amongst all eight cores also suggests that location B was subject to more sand incorporation than location A due to the wide discrepancy in K_{sat} values irrespective of core orientation.

Table 5.1. K_{sat} values from horizontally and vertically sampled clay cap (June, 1986).

Location	Core Orientation	Replicate No.	Texture*	Dry Bulk Density	K_{sat}
				$g \cdot cm^{-3}$	$cm \cdot s^{-1}$
A	horizontal	1	sicl	1.61	4.0×10^{-7}
		2		1.59	4.5×10^{-7}
	vertical	1	c	1.60	1.0×10^{-7}
		2		1.55	5.1×10^{-8}
B	horizontal	1	cl	1.56	1.5×10^{-4}
		2		1.60	5.2×10^{-6}
	vertical	1	cl	1.53	1.9×10^{-5}
		2		1.60	2.6×10^{-5}

*sicl-silty clay loam; c - clay; cl - clay loam

The procedure described above was repeated later in the season (August) at two randomly selected locations in the northern sector of the landfill area to further evaluate the influence of sand intermixing and possibly clay volumetric shrinkage (cracking) on K_{sat} values. In addition, particle size analyses (i.e. pipette method with sand fractionation) were performed and all data are presented in Table 5.2. There appeared to be less visible and analytical evidence of sand intermixing at locations C and D with the texture ranging from silty clay to silty clay loam. Both the sand and clay contents of the cap were higher at location D, thus indicative of appreciable natural variation in the clay material texture without the compounding problem of local sand incorporation. Widespread visible clay cracking was not in evidence in these eight cores or at sampling locations C and D in general. Nevertheless, the textural similarity to the capping material at location A (Table 5.1), the somewhat lower dry bulk densities in August, and the absence of any K_{sat} measurements in the 10^{-7} to 10^{-8} $cm \cdot s^{-1}$ range strongly suggest that significant soil structural influences on water transmissibility have developed during the summer months.

A further investigation was undertaken into the permeability of the clay landfill cover using several tensiometer nests placed within, 5-10 cm below and 10-15 cm below the clay layer. From this it was determined that the clay cap was generally adequate in reducing infiltration during rainfall events where the depth of capping material exceeded 10 cm and the degree of sand mixing was low. The lack of appreciable change in negative pressure potential measurements taken during and soon after precipitation events in these capped areas (i.e. silty clay loam to clay material greater than 10 cm deep) suggests that surface infiltration is minimized on these portions of the landfill.

No data on surface runoff versus infiltration volumes were collected in 1986 but field observations were made during and subsequent to any major storms when one or more members of the research team were present at the site. Within the eastern sector of the landfill (Figure 5.1), little runoff if any was observed implying near complete infiltration. In contrast, appreciable runoff was observed on the sloping western sector of the landfill surface and has resulted in extensive gullying of not only the capping material but also the underlying refuse. This suggests that both the inherent erodibility of the capping material and the slope configuration of the landfill surface are crucial to the design of a secure landfill.

Table 5.2. K_{sat} values and particle size distribution from horizontally and vertically sampled clay cap (August, 1986).

Location	Core Orientation	Replicate No.	Particle Size Distribution: Gravel	vcs	cs	ms	fs	vfs	Total Sand	Total Silt	Total Clay	Textural Class	Dry Bulk Density	K_{sat}
			$\% \; g \cdot g^{-1}$										$g \cdot cm^{-3}$	$cm \cdot s^{-1}$
C	horizontal	1	0.0			n.a.			5.1	57.6	37.3	sicl	1.50	3.4×10^{-5}
		2	0.0			n.a.			6.4	56.9	36.7	sicl	1.50	4.2×10^{-6}
	vertical	1	0.2			n.a.			6.4	54.4	39.2	sicl	1.52	3.0×10^{-5}
		2	0.0			n.a.			4.9	53.2	41.9	sic	1.47	5.5×10^{-4}
D	horizontal	1	0.0	0.3	1.0	1.9	3.5	6.2	13.0	41.3	45.7	sic	1.53	7.7×10^{-6}
		2	0.0	0.2	0.7	2.1	3.4	7.4	13.8	43.5	42.7	sic	1.55	1.7×10^{-5}
	vertical	1	0.0	0.1	0.3	1.3	2.3	7.9	11.9	44.5	43.6	sic	1.43	1.2×10^{-6}
		2	0.0	0.3	0.9	2.7	5.0	10.2	19.2	41.8	39.0	sicl	1.56	1.2×10^{-6}

n.a. - not applicable (i.e. sand fractionation is not performed when the total sand is less than 10% $g \cdot g^{-1}$)

sicl - silty clay loam; sic - silty clay; vcs - very coarse sand; cs - coarse sand; ms - medium sand; fs - fine sand; vfs - very fine sand

5.2.2 Alternative Capping Materials

From the aforementioned investigations, it was apparent that alternative natural and synthetic capping materials should be studied with a view to recommending a cost-effective liner for consideration by the Municipality. More substantive data were also required to more fully evaluate the infiltration characteristics of the existing cap material. To this end, a series of nine 1 m^2 plots were established on the sloping western sector of the landfill. The relative efficiencies of several materials and soil treatments situated under a 5 cm layer of native turf in reducing or eliminating downward percolation of post-infiltration soil water were assessed. The treatments included the local sand (compacted and uncompacted) and the existing clay cap, a bentonite slurry with and without sand incorporated, a bentonite-polyvinyl alcohol (PVA) mixture, an industrial polymer latex-sand mixture and plastic. A compacted clay cap treatment was also initially included but was later discounted due to an apparent depth and textural inconsistency with its uncompacted counterpart. Physical barriers imbedded to a depth of about 20 cm were used to segregate the plots on three sides, leaving the downslope side open for collection of surface runoff and interflow above the treated layer.

Rainfall events were simulated and the volume of water applied to each plot while at field capacity (i.e. plots pre-saturated and drained) was compared to the volume recovered. A simple moisture balance was used to determine the volume of incident water lost to deep drainage beyond the vegetative root zone.

The results of one such trial are presented in Table 5.3 and are representative of other trials performed. The results for September 20 indicate that the industrial polymer latex-sand mixture and the bentonite slurry approach the total exclusion of water that the plastic liner affords while allowing sufficient deep drainage to promote microbial decomposition of the landfill refuse. All other treatments fell well short of this optimum degree of water transmissibility which was arbitrarily defined as runoff exceeding deep drainage by a factor of two or more. The existing clay cap was the least effective treatment in reducing deep drainage, even relative to the local sand material. The volumetric shrinkage of the predominantly illitic clays upon drying is thought to be sufficient to cause some cracking through the full depth of the thin clay material. These structural macropores are capable of increasing the effective hydraulic conductivity of cracked clays by several orders of magnitude to in excess of even sand materials.

Table 5.3. Results of the soil moisture balance for a single irrigation event on the alternative landfill cap plots.

Soil Treatment	Soil Moisture Balance for Irrigation Event - Sept.20/86 Depth of Irrigation	-	Actual Evapotranspir-ation	=	Runoff Collected	+	Deep Drainage
	mm		mm		mm		mm
Plastic	8.10		2.43		5.67		0
Polymer latex-sand	8.10		2.43		4.25		1.42
Bentonite slurry	8.10		2.43		4.66		1.01
Bentonite - PVA	8.10		2.43		3.14		2.53
Bentonite - sand	8.10		2.43		2.84		2.83
Existing clay cap	8.10		2.43		1.82		3.85
Sand (control)	8.10		2.43		2.73		2.94
Compacted sand	8.10		2.43		2.94		2.73

This analysis suggests that the industrial polymer latex, which is currently available at no cost, is a particularly attractive alternative to some of the more expensive liners now being used at some engineered landfill sites outside of Ontario. Examples include admixed liners (soil cement, soil asphalt), flexible polymeric membranes (Dupont Hypalon, polyethylene) and sprayed-on linings (U.S.E.P.A., 1980). The precise composition of the latex is unavailable at this time but Table 5.4 presents the results of a cursory chemical analysis. There do not appear to be any inorganic substances which might be of environmental concern should some weathering occur of a surface liner comprised of this material.

Table 5.4. Analysis of the industrial polymer latex.

Element	Concentration	Element	Concentration
	% $g \cdot g^{-1}$	(transition metals)	$mg \cdot kg^{-1}$
N	0.0	Ni	< 0.1
P	0.01	Cd	< 0.1
K	0.0	Zn	1.2
Mg	0.0	Co	< 0.1
Ca	0.03	Pb	< 0.1
		Fe	4.0
		Mn	< 1.0
		Cr	< 0.1
		Cu	< 0.5

In addition to the analytical results in Table 5.4, it is known that the latex material is a suspension of vinyl acetate, butyl acrylate and ammonium persulfate in deionized water. It is the latter substance which decomposes and releases free radicals, thus causing the monomer mixture to

polymerise into long chain molecules (A. Roberts, pers. comm.). Once dried, a high strength veneer of low permeability is created.

Two latex test plots were established on a slope at the base of the landfill cap late in the 1986 field season. The plots (1 m x 10 m) were prepared by removing all traces of vegetation and the subsurface clay material. Both sand plots were covered with a dilute mixture of latex. On one plot, a mortar-like paste of pre-mixed latex, water, sand and clay (i.e. 6:6:28:1 by volume) was evenly applied to create a 5 cm thick veneer. The second plot was treated by pouring 100 ℓ (i.e. 10 $\ell \cdot m^{-2}$) of latex onto the surface and mixing it with the sand material by raking. The sloping portion of the test plots proved to be the most difficult to work on due to slumping but the result was a sheet of polymer latex covering both plots. The plots were left to "weather" over the winter months and are being tested at the time of writing for permeability, erodibility and trafficability. Initial results indicate that the integrity of both plot liners was effectively unaltered by exposure to the elements.

5.3 Reduction of Groundwater in Contact with Landfill Refuse

5.3.1 Subsurface Hydrology of the Site

5.3.1.1 Preface

As stated previously in the report, of paramount importance to the long-term resolution of the pollution hazard at Muskoka Lakes is the reduction of the volume of leachate generated. To this end, a hydrogeological evaluation of the site was conducted to build upon the findings from earlier reports. The investigation included the installation of a piezometer grid in the vicinity of the landfill and monitoring of the potentiometric surface to determine groundwater flow direction and velocity. Localized use of fluorescent dyes was intended to isolate local groundwater flow systems. A further investigation was undertaken to delineate leachate concentration isolines to advance the characterization of the leachate plume. In addition, the feasibility of dewatering the area beneath the landfill using a single well was examined. This might eliminate any mounding of the groundwater table in the refuse body and possibly provide a more effective leachate collection system than is currently in place.

A. Roberts, Reichhold Ltd., Weston, Ontario.

5.3.1.2 Groundwater Flow Network

A close inspection of low altitude aerial photographs revealed that the landfill site is located in a bedrock trough which is orientated in essentially a north-south direction. Surface runoff and interflow originating from the north are diverted by a natural ravine, whereas from the west these are intercepted by a perennial stream which flows south to Lake Muskoka. An outcrop ridge situated upgradient and adjacent to the landfill further diverts both surface and groundwater flow originating from the east around the landfill.

An extensive piezometer network was established early in the study to determine groundwater levels in the vicinity of the landfill. Initially, a total of 28 piezometers were installed, the locations of which are shown in Figure 5.3. The piezometers consisted of 3/4" PCV pipes fitted with sand screens and were installed to depths of at least 1 m below the groundwater table or to bedrock. A water level meter was used to monitor the piezometers bi-weekly.

It is evident from the flow net configuration that, like the surface runoff and interflow, the groundwater flow originating from the north discharges into the natural ravine and subsequently flows toward the perennial stream, thus circumventing direct contact with the landfill. Moreover, the flow net establishes that groundwater flow from the landfill itself is directed west toward the collection trench. Analysis of the potentiometric surface, however, neither definitively confirmed nor eliminated the possibility that a bedrock ridge or some other obstruction to groundwater flow might be situated between the landfill and the collection trench. To further investigate this possibility, six additional piezometers were installed at the base of the landfill slope in August 1986. As illustrated in Figure 5.4, the additional data confirmed a predominantly westerly flow to the collection trench, apparently unobstructed.

A related investigation involved installation of three open standpipes within, and immediately adjacent to, the collection trench. The purpose was to measure the pressure potential at different depths below the soil surface, thereby establishing whether or not positive pressures existed in the vicinity of the trench which were large enough to cause upward vertical flow (i.e. discharge of a confined aquifer). The results indicated that no appreciable vertical pressure gradient existed that would induce upward flow to the collection trench. It is, therefore, likely that the level in the

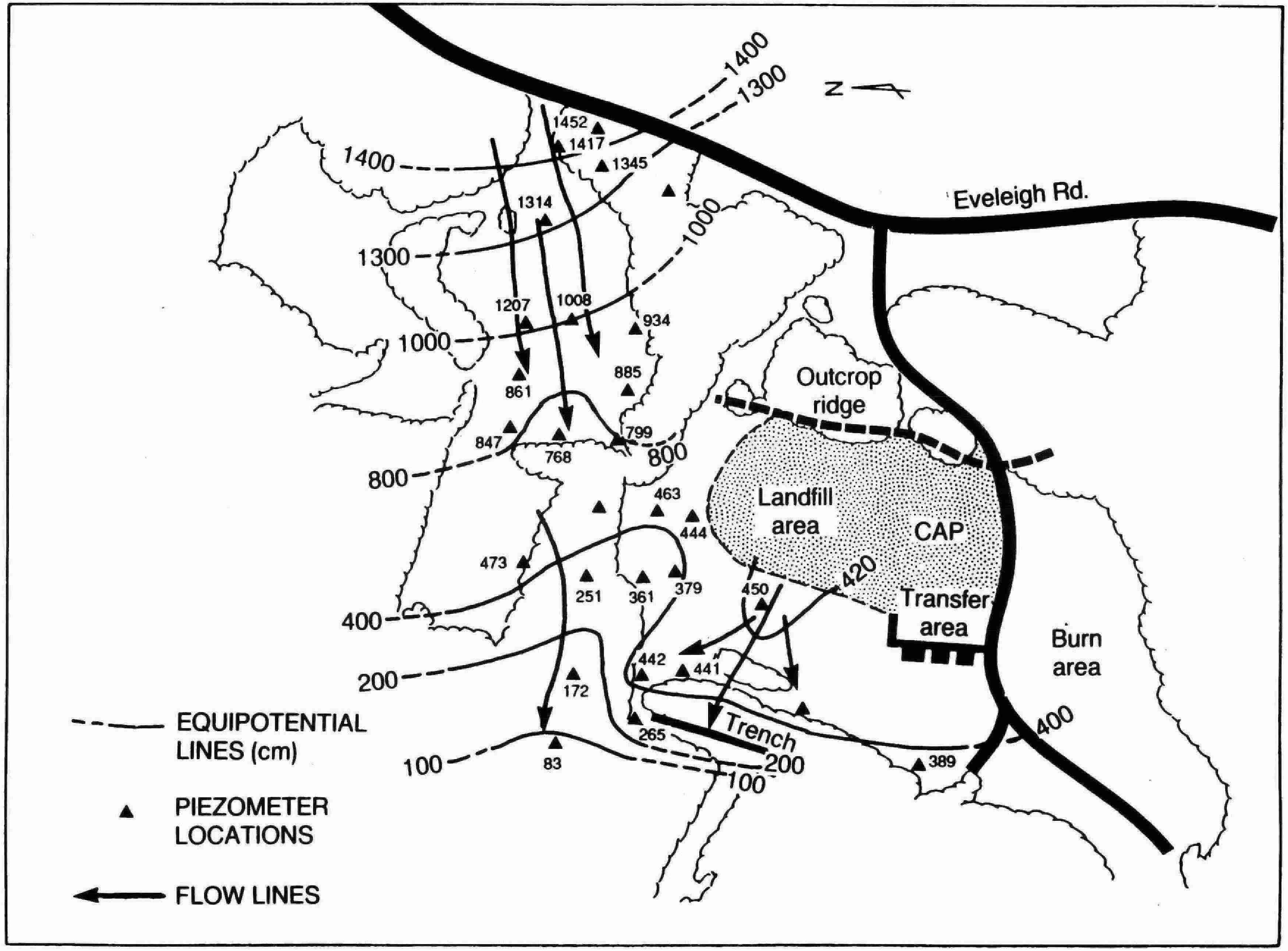


Figure 5.3. Groundwater flow network (June, 1986) (scale 1:3,900).

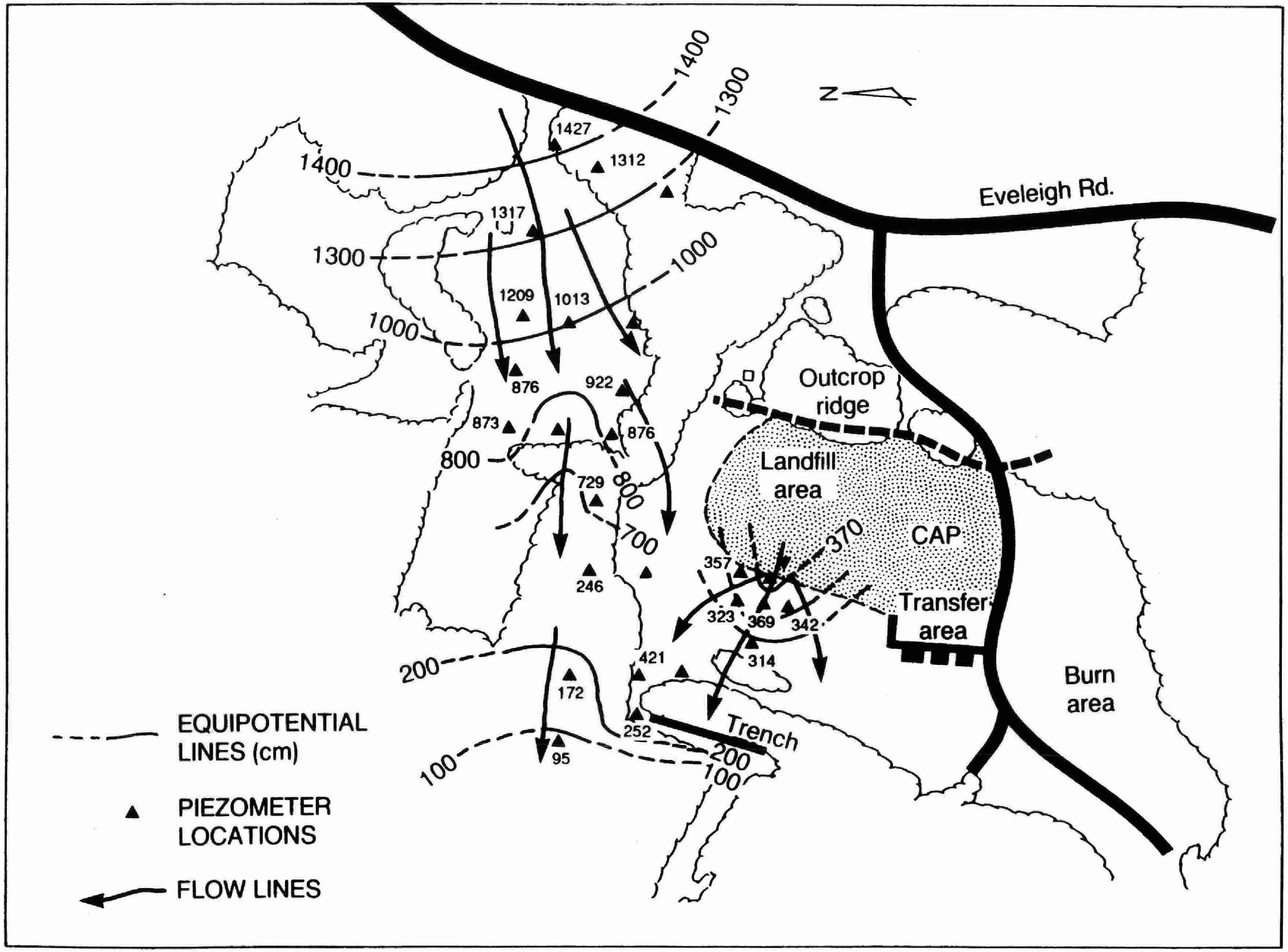


Figure 5.4. Groundwater flow network (October, 1986) (scale 1:3,900).

collection trench is largely maintained by seepage from a nearby embankment and no confined aquifer exists.

Attempts were also made to monitor the migration of fluorescent dyes in local flow systems to further characterize groundwater flow directions and velocities. Uranine (yellow) dye was added to surface pools upgradient from the landfill in order to more definitively establish the flow patterns originating from the east of the landfill where no piezometers were installed. Rhodamine B (red) dye was also injected below the groundwater table in the sand pit area to the northwest of the landfill. This was done to confirm the linkage of subsurface flow from ephemeral pools in the sand pit area to the leachate collection trench. Surface water was frequently sampled at twelve pre-selected monitoring stations and were carefully inspected under an ultraviolet light source. At no time were traces of fluorescent dyes observed in any of these samples. Consequently, no flow velocities or directions were directly ascertained from this investigation.

There are several possible explanations for the lack of success of the dye study: i) divergent flow patterns diluted the dyes below detectable limits, ii) the groundwater flux was underestimated to the extent that insufficient dye was used, iii) flow velocities were overestimated implying that the monitoring time was insufficient, and iv) organic substances in the soil or groundwater absorbed or denatured the fluorescent dyes. The reason most frequently cited by researchers for failed dye monitoring trials, however, is error in prior estimation of the flow net leading to improper monitoring station placement.

5.3.1.3 Aquifer Decontamination

The Contaminant Plume

A Geonics EM31 electromagnetic induction meter was employed to map contaminant concentration isolines between the landfill and the collection trench. Greenhouse and Harris (1983) cite two important criteria that must be satisfied when employing this non-contact electromagnetic ("inductive") conductivity instrument. First, the introduction of contaminants into the aquifer must result in a quantifiable change in the total electrical conductance of the volume of earth sensed by the instrument (i.e. 6 m depth for the EM31 in a "vertical" position). Secondly, the lateral variations in electrical conductivity due to changing lithology, or "natural scatter" (Klefstad et al., 1975), must be distinguishable from those due to groundwater contamination. As apparent on the isopleth maps in Figures 5.5

and 5.6, not only was the introduction of contaminants to the aquifer measurable but also the general pattern of contaminant concentration isolines was reproducible in June and September 1986. Moreover, due to the relatively homogeneous stratigraphy in the vicinity of the landfill, natural scatter levels in a nearby uncontaminated area were ascertained prior to mapping groundwater contamination and found to be negligible, thus eliminating a possible source of variance caused by lithological factors. Indeed, the Precambrian bedrock and highly leached sand overburden exhibited electrical conductivity levels below detectable limits with this instrument.

Closer inspection of the contaminant plume in Figures 5.5 and 5.6 reveals that the zone of highest apparent electrical conductivity (ECa) tends to vacillate somewhat in its position by up to 20 m. Speculation that a pool of highly concentrated leachate might be migrating beyond the limit of the landfill each spring toward the perennial stream was quashed by a third EM31 scan in March, 1987. The resulting isopleth map showed that this concentrated zone of leachate had not migrated any closer to the collection trench since the October scan.

In addition to identifying the area of greatest electrical conductivity (i.e. highest leachate concentration), the investigation also confirmed the direction of leachate flow as being from the landfill toward the collection trench with no evidence of aquifer contamination beneath the sand pit area immediately to the northwest.

Pump Testing

The feasibility of dewatering the area beneath the landfill using a single pumping well was further examined. By lowering the groundwater table and maintaining it at reduced levels, it might be possible to eliminate any mounding of the zone of saturation within the refuse body. This, coupled with the installation of a suitable landfill surface liner, would give rise to a marked decline in leachate strength and volume over time. Inducing a widespread drawdown within the contaminated aquifer might also prove to be a more effective and complete means of collecting the leachate as compared to the present shallow trench system.

The occurrence of groundwater table mounding within or beneath landfilled wastes has been reported by numerous researchers (Freeze and Cherry, 1979; MacFarlane et al., 1983). Where refuse has been placed in contact with or in proximity to the phreatic surface, the zone of saturation

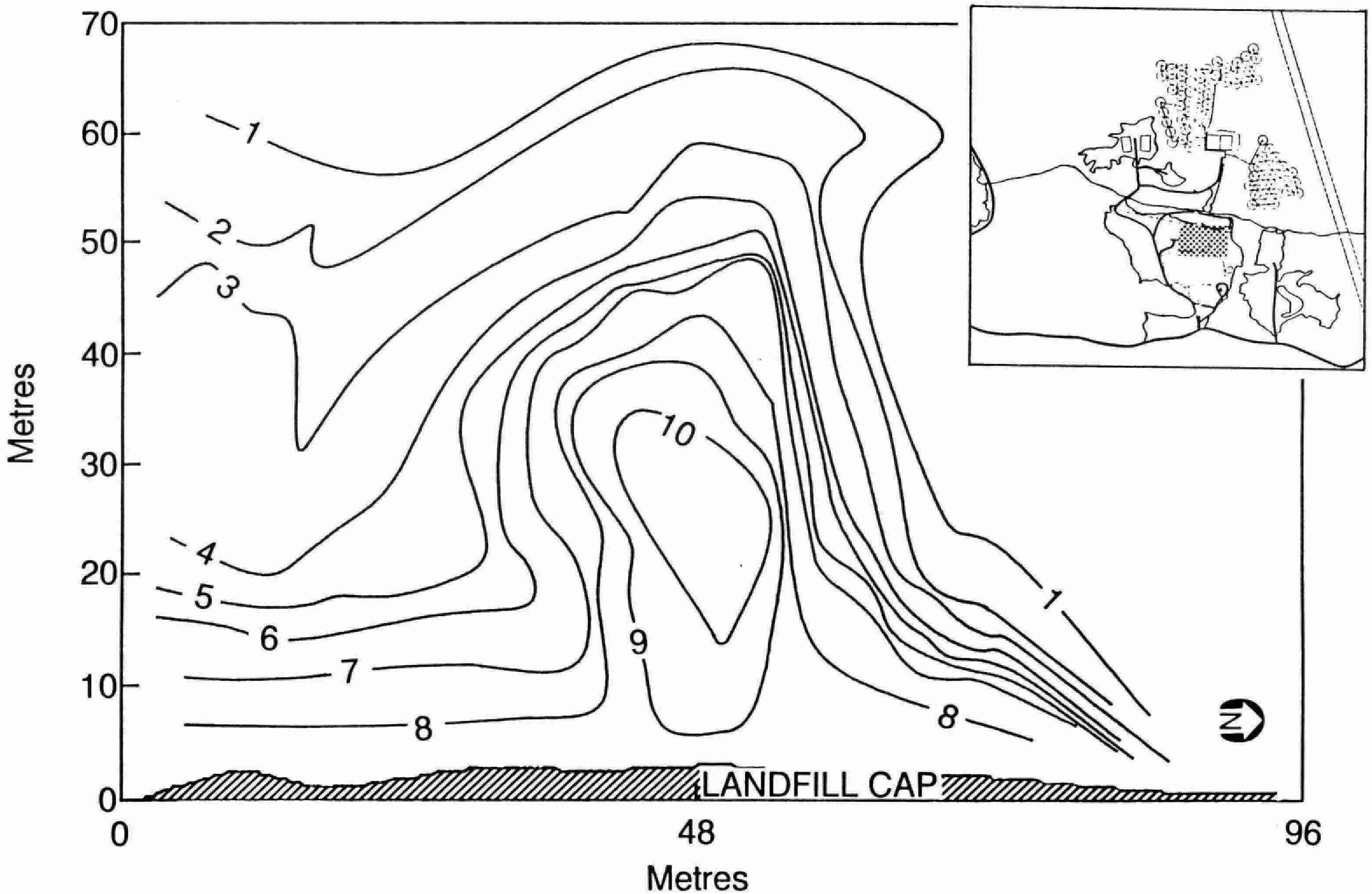


Figure 5.5. Isolines of apparent electrical conductivity (ECa in mmhos$\cdot$m^{-1}) between the landfill and the collection trench (May, 1986).

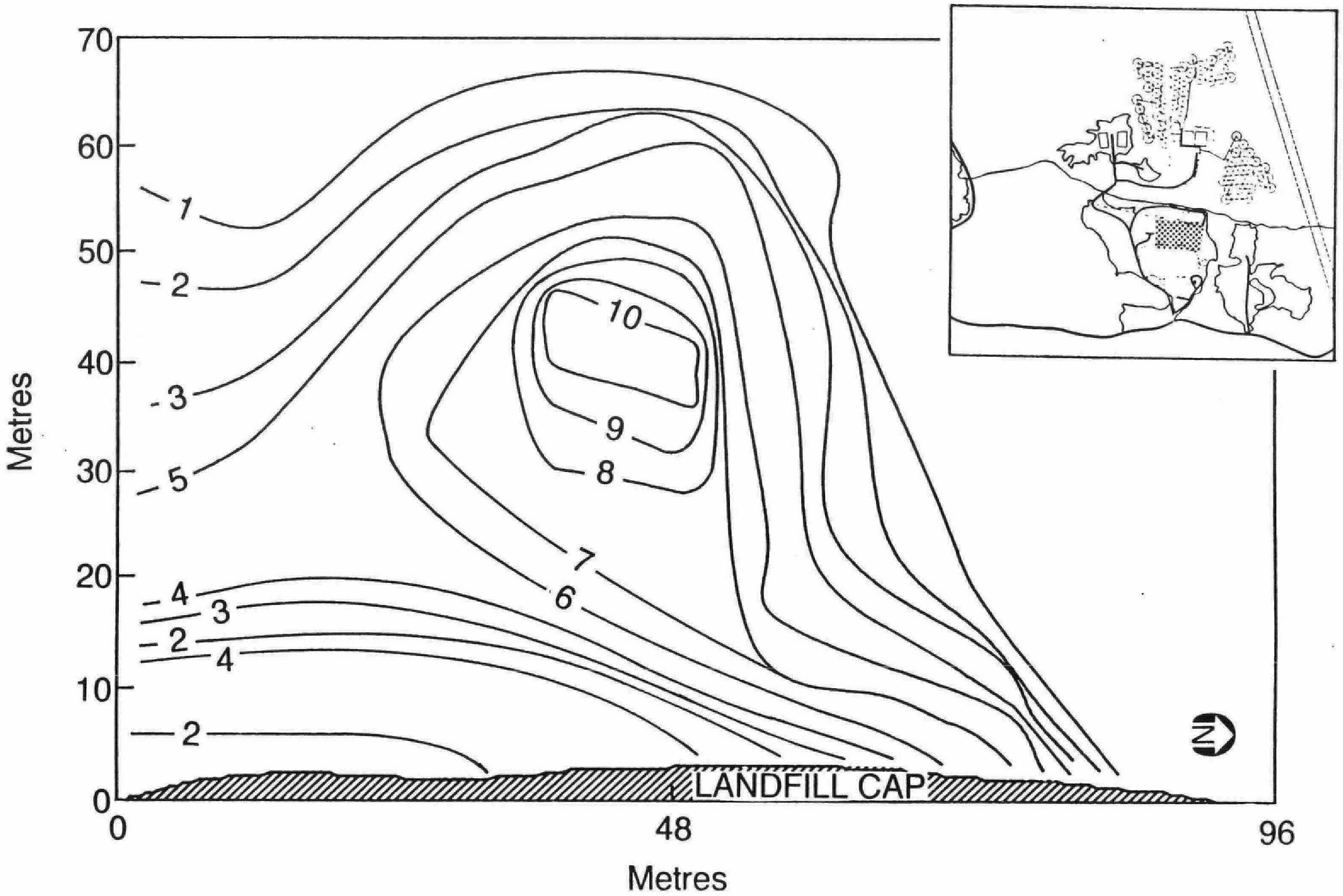


Figure 5.6. Isolines of apparent electrical conductivity (ECa in mmhos$\cdot$m^{-1}) between the landfill and the collection trench (September, 1986).

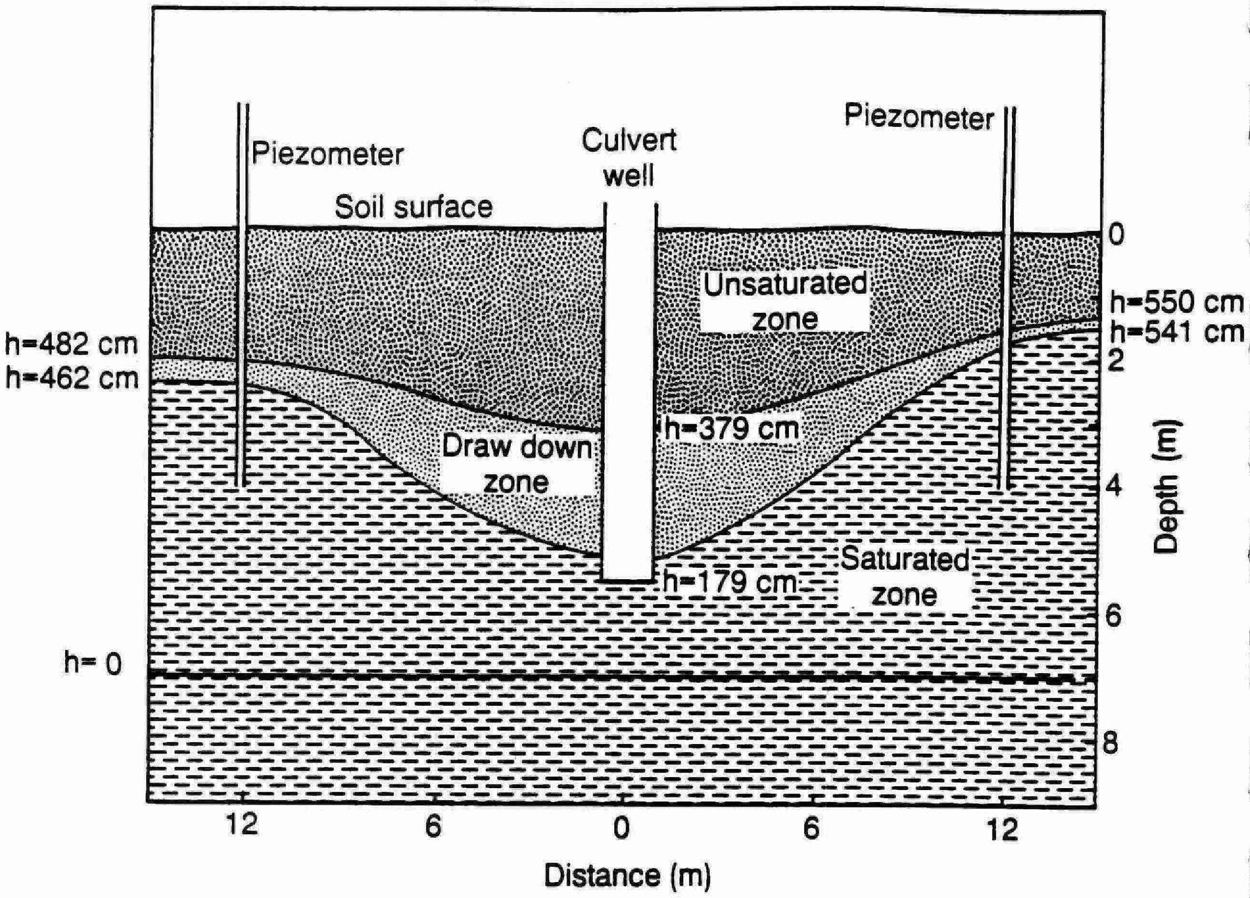
tends to mound vertically resulting in increased leachate volume, strength and lateral migration (Straub and Lynch, 1982a; 1982b).

Two steel piezometers were installed within the landfill itself in June, 1986 in order to confirm or refute the presence of a groundwater mound. One was drilled to a depth of about 6 m near the centre of the landfill while the second was installed to just over 8 m in depth further west where the slope contour becomes more pronounced. Obstructions encountered at the 6 and 8 m depths, respectively, made it impossible to reach the bedrock at the base of the refuse body. Frequent soundings were taken between mid-June and October but at no time was a free water surface encountered in either piezometer. These results were not entirely unexpected as groundwater mounds have been reported to obtain their maximum elevation in the spring and subside during summer, fall and winter months. (MacFarlane et al., 1983)

Once it was known that the vertical extent of any mounding was not sufficiently large to make pumping impractical, a culvert well (6 m x 72 cm) was installed at the base of the landfill to test the feasibility of dewatering the contaminated aquifer. The culvert well was situated in the centre of the area previously identified by the Geonics EM31 in May as having the greatest leachate strength, or at least comprising the largest volumetric pool according to the measured apparent electrical conductivity (ECa) (Figure 5.5). In addition, piezometers were positioned at four points 12 m away from the culvert well to monitor the effect of pumping on the phreatic surface. A pump test was conducted and resulted in a drawdown at 12 m ranging from 9 to 20 cm after only eight hours of maintaining a level in the well 2 m below the initial phreatic surface (Figures 5.8a and 5.8b). The h = 0 datum represents the level of the perennial stream.

Following the pump test, the subsequent recharge and drawdown reversal were monitored via the culvert well and surrounding piezometers, respectively, to determine the time required to reach the original phreatic surface level. Using a procedure reviewed by van Beers (1963), the field saturated hydraulic conductivity (K_{fs}) was estimated from the recharge data. According to this model, the K_{fs} is approximately 5.0×10^{-4} cm$\cdot$s^{-1}. This rate is over one order of magnitude slower than results obtained using the Guelph Permeameter in the local soils (see section 6.4.7). This discrepancy was anticipated due to the limited number of inflow ports cut into the side of the culvert well before installation. It is believed that the

a)



b)

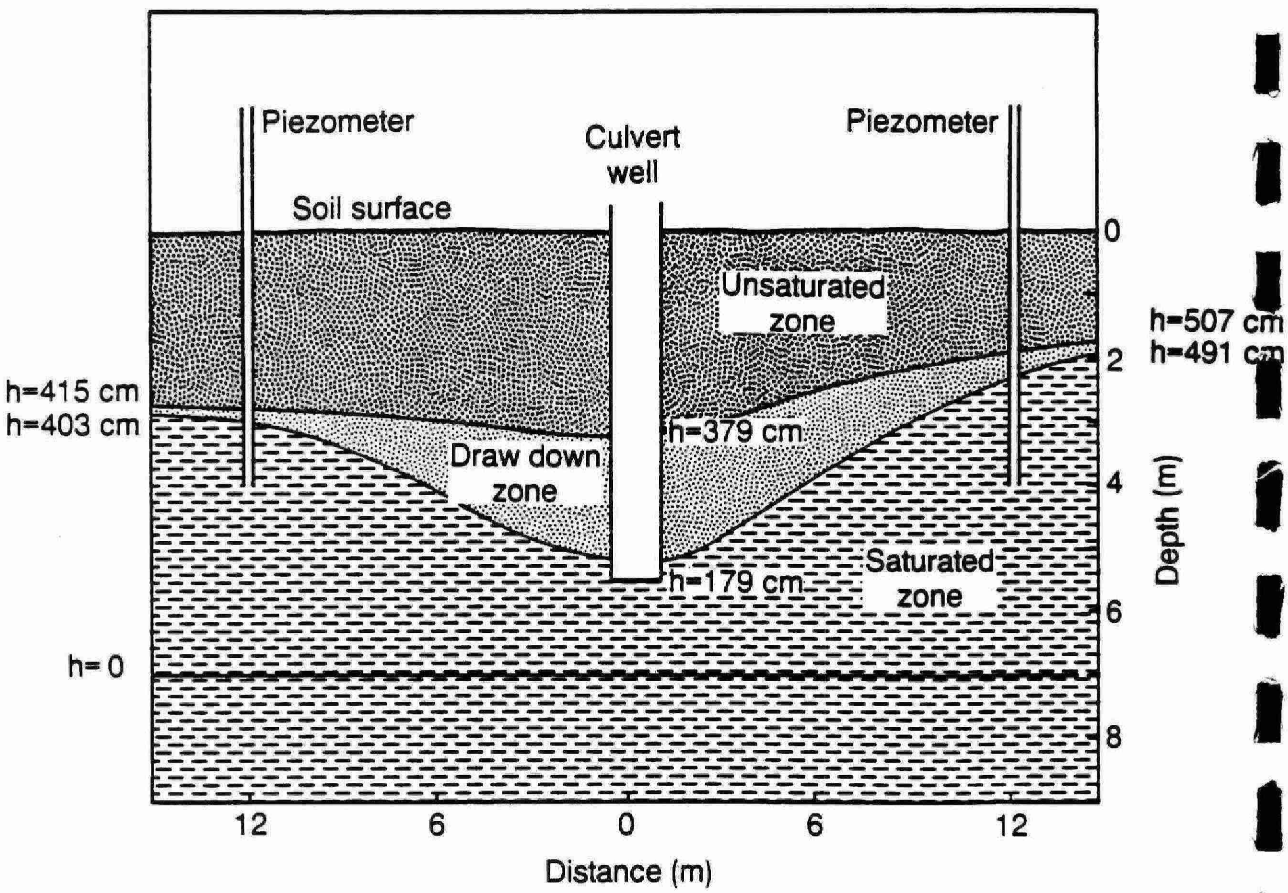


Figure 5.7. Drawdown zone around the culvert well after a pumping test of 8 hours duration a) east-west piezometer line b) north-south piezometer line.

"permeability" of the culvert well wall was thus the overriding limiting factor in establishing water flux into the well using this variant of the auger hole method for K_{fs} measurement.

Figures 5.3 and 5.4 suggest that the hydraulic potential gradient in the landfill vicinity is about 0.03, and thus considerably lower than the earlier M.O.E. (1975) estimate of 0.12. Using Darcy's Law and a K_{fs} of 7.5 x 10^{-3} $cm \cdot s^{-1}$ (i.e. as per the Guelph Permeameter measurements in section 6.4.7), the predicted migration would be about 70 $m \cdot yr^{-1}$.

5.4 Conclusions

From hydrologic investigations carried out at the Muskoka Lakes landfill in 1986, it is believed that both the leachate volume and strength can be dramatically reduced from current levels which would augur well for the prospects of continued spray irrigation operations at this site. This would best be achieved through a reduction of net infiltration though the landfill surface coupled with a reduction in the volume of groundwater in direct contact with landfilled wastes.

Major groundwater diversion works are not believed to be necessary at this site due to the natural hydrologic isolation of the landfill, particularly to the north and east. Occurrences of springs beneath the landfill, which could arise from rapid secondary permeability along bedrock fracture lines, are seemingly not a factor at this site. This premise is supported by on site geological observations as well as the correspondence between rainfall amounts incident on the microwatershed which encompasses the landfill and the volume of leachate collected (see section 6.2).

The groundwater flow characteristics in the landfill vicinity have been more precisely defined. The permeability of the indigenous soil is about 7.5 x 10^{-3} $cm \cdot s^{-1}$, although local variations may occur due to coarser textured substrata. The hydraulic potential gradient is approximately 0.03 which results in an anticipated rate of groundwater migration of about 70 m annually. Flow directions on the east side of the perennial stream are predominantly toward the west as found in earlier consulting reports.

The existing landfill cap cannot be regarded as an appropriate surface liner over the long-term. As an interim surface treatment, however, the clay is suitably impermeable where it is greater than 10 cm thick and where intermixing with local sand materials has been avoided. As present, this constitutes no more than 22% of the total landfill area. Even these areas

offer only marginal protection of the landfilled wastes from incident rainfall over the long-term due to i) the inherent erodibility of the silty clay to silty clay loam materials, and ii) the appreciable slope length and contour gradient of the landfill surface. It is the lack of continuity of even these marginally suitable cap conditions which largely negates any benefit accruing from the emplaced clay. Much of the surface runoff from the better sealed areas will eventually encounter a cap discontinuity, infiltrate the surface, and become deep drainage within the refuse body. Alternative solutions include the installation of a synthetic plastic surface liner or the importation of large quantities of clay material from an even more distant source of fill material (i.e. up to 1 m clay depth required on landfill), both of which may prove prohibitively expensive. A third alternative might involve application of a low cost industrial polymer latex material which on site studies have shown possesses the necessary water transmissibility characteristics. Good success has been achieved in tests of rubber and plastic latexes when applied as a spray-on liner for waste impoundments in the U.S. (U.S.E.P.A., 1980). More investigations are ongoing into the weatherability, trafficability and erodibility of this material which are necessary before a final recommendation can be made.

A well defined contaminant plume has been delineated between the landfill and the collection trench. There are no indications of contaminant diffusion to the north (i.e. the sand pit area) but the plume is apparently widespread to the south due to the influence of contaminants introduced from the dumping and burning activities in that area. A pump test demonstrated the feasibility of dewatering the contaminated aquifer by continuous pumping from the culvert well to the sump pump. It is believed that aquifer dewatering will eventually curb the seepage occurring along the embankment immediately to the east of the collection trench and eliminate any groundwater mounding that may be occurring in the landfilled wastes. This will lower the strength of the leachate with time as the groundwater is removed from direct contact with the wastes.

6.0 Study of Soil Quality for Leachate Disposal

6.1 Preface

The present spray area of 4.3 ha is composed of a veneer of rapidly permeable fine and medium sands (occasional coarse sand or gravelly substrata) with depths to bedrock ranging from 0 m (outcrops) to greater than 3.4 m (Gartner Lee Assoc. Ltd., 1984). Groundwater flow is controlled by the bedrock relief and the flow systems are thus more local than regional in character and have been disrupted in the landfill site area. The recommended design application rates of 23,000 $\ell \cdot ha^{-1} \cdot d^{-1}$ (April, May, September, October) and 34,000 $\ell \cdot ha^{-1} \cdot d^{-1}$ (June, July, August) were based on a very simplistic water balance with little regard for soil infiltration rates, water storage capacities, attenuation capacities, depth to bedrock, or the composition of the leachate. Deciduous tree species and forage grasses have been shown to withstand leachate irrigation rates of up to 6.5 $mm \cdot d^{-1}$ from October to April in West Virginia, but under very different soil conditions than those at the Muskoka Lakes site (Menser et al., 1979). Acceptable loadings in the U.K. are less at about 5.6 $mm \cdot d^{-1}$ (Harrington and Maris, 1986).

The purpose of this segment of the study was to establish the attenuation capacity of the local soil materials both in situ and in a laboratory environment. Of equal importance was consideration of the extent to which the suspended organic particulate content of the leachate and the chemical precipitation of inorganic contaminants in the soil might contribute to a general decline in surface soil infiltration rate and attenuation capacity with time. More quantitative estimates of the volumes of leachate currently being generated and, therefore, of the necessary spray application rates in the hardwood forest are made in section 6.2. Chemical changes in the raw leachate over time (1978-86) and at various points along the collection/disposal system are further summarized in section 6.3. These data are critical to the interpretation of the soil matrix and soil solution analyses to follow. Both sprayed and unsprayed soils were fully characterized in terms of standard chemical and physical analyses and prevailing water regimes (section 6.4). Groundwater and vadose water quality, given the present spray regime, were also monitored along major bedrock-controlled flow lines leading to break-out areas and the perennial stream (section 6.4). Soil leaching columns were employed in the laboratory to determine the attenuation and elution patterns for several key leachate constituents under variable application rates and forms of physico-chemical

pretreatment (section 6.5). Together with data on evapotranspirational demand, the above information would allow refinement of recommended leachate application rates.

In winter, excess leachate from the collection trench presently overflows directly into the perennial stream which in turn drains into the North Bay of Lake Muskoka. Sub-irrigation or other slow rate infiltration techniques might allow leachate to be treated on land during the winter. A preliminary assessment of the alternative distribution methods was initiated in 1986 for this purpose (section 6.5).

6.2 Leachate Volume Generation and Corresponding Land Application Rates

Ideally, spray application rates of leachate on land should be attuned to atmospheric evapotranspirational demand, the composition of the wastewater, the capacity of the local vegetation to withstand leachate exposure and the local soil conditions (e.g. infiltration rate, water storage capacity, attenuation capacity, depth of bedrock). In practice, however, these considerations are often by necessity superseded by the excessive volumes of leachate being collected during certain times of the year relative to the capacity of the holding lagoon(s) used to "balance" the leachate strength and flow variations (see section 3.4.2.1).

To more accurately define the volume of leachate being collected by the existing shallow trench system, a cumulative timer was placed on the sump pump in mid-July and periodically monitored. In addition, the sump's pumping capacity was determined to be approximately 1.64 $\ell \cdot s^{-1}$. Table 6.1 contains information on the cumulative pumping times and estimated leachate volumes pumped to the settling lagoons for three periods. The estimated daily leachate volumes increase very rapidly from late July to late September which is reflective of the varying precipitation regimes during this two month period (see section 4.0). For the 63 day period that the sump timer was monitored, the total volume pumped was about $3.34 \times 10^6 \ell$, which aggregates to an annual total of $1.94 \times 10^7 \ell$. This value is significantly lower than the $2.4 \times 10^7 \ell \cdot yr^{-1}$ estimate made by Gartner Lee Assoc. Ltd. (1984), recognizing that all months during the spray season received higher than normal precipitation amounts in 1986 (see section 4.0). This might suggest that the present shallow trench collection system is becoming less effective with time in intercepting flow from the contaminated aquifer due to sludge sedimentation or other factors.

Table 6.1. Cumulative pumping times and estimated leachate volumes for three periods.

Period	Cumulative Pumping Time	Est. Total Leachate Volume Collected	Est. Daily Leachate Volume Collected
	s	ℓ	$\ell \cdot d^{-1}$
July 24-31	1.3×10^5	2.13×10^5	2.67×10^4
Aug.1-Sept.1	1.0×10^6	1.64×10^6	5.29×10^4
Sept.1-Sept.24	9.1×10^5	1.49×10^6	6.21×10^4

It had been noted through the course of the field season that there was a discernable lag between major precipitation events and the resulting increase in leachate flow to the collection trench. To better define this phenomenon, both daily precipitation amounts and the sump timer data were plotted in Figure 6.1. The approximate lag or response time was then determined from the number of days between a heavy rainfall and the corresponding increased timer response (e.g. a - a'). Although some variation was observed, a lag time of about seven days was clearly indicated. It is believed that this delayed flush originates largely from rainfall incident on the sand flats between the landfill and the trench. The highly contaminated leachate originating from precipitation passing through the landfill provides the "baseflow" to the trench since it migrates over much greater distances and hence is not manifested in peak flows.

Data from the sump timer also allowed for a cursory assessment of the relative contributions of incident precipitation and migrating groundwater from beyond the immediate vicinity of the landfill to leachate volumes. A micro-watershed was delineated which encompassed the landfill and the sand flats to the west and had an estimated area of 3.3 ha. It was assumed that precipitation incident on this area would migrate via surface runoff, interflow or groundwater flow to the collection trench. During the month of August, 96 mm of precipitation fell on this area which amounts to $3.2 \times 10^6 \ell$ of water. Based on the seven day lag period established above, the sump timer indicated a total pumping time of 1.11×10^6s from August 7 to September 7 which corresponds to $1.82 \times 10^6 \ell$ of leachate pumped. Assuming that i) a significant proportion of the incident precipitation is lost to evapotranspiration, and ii) the shallow trench collection system does not

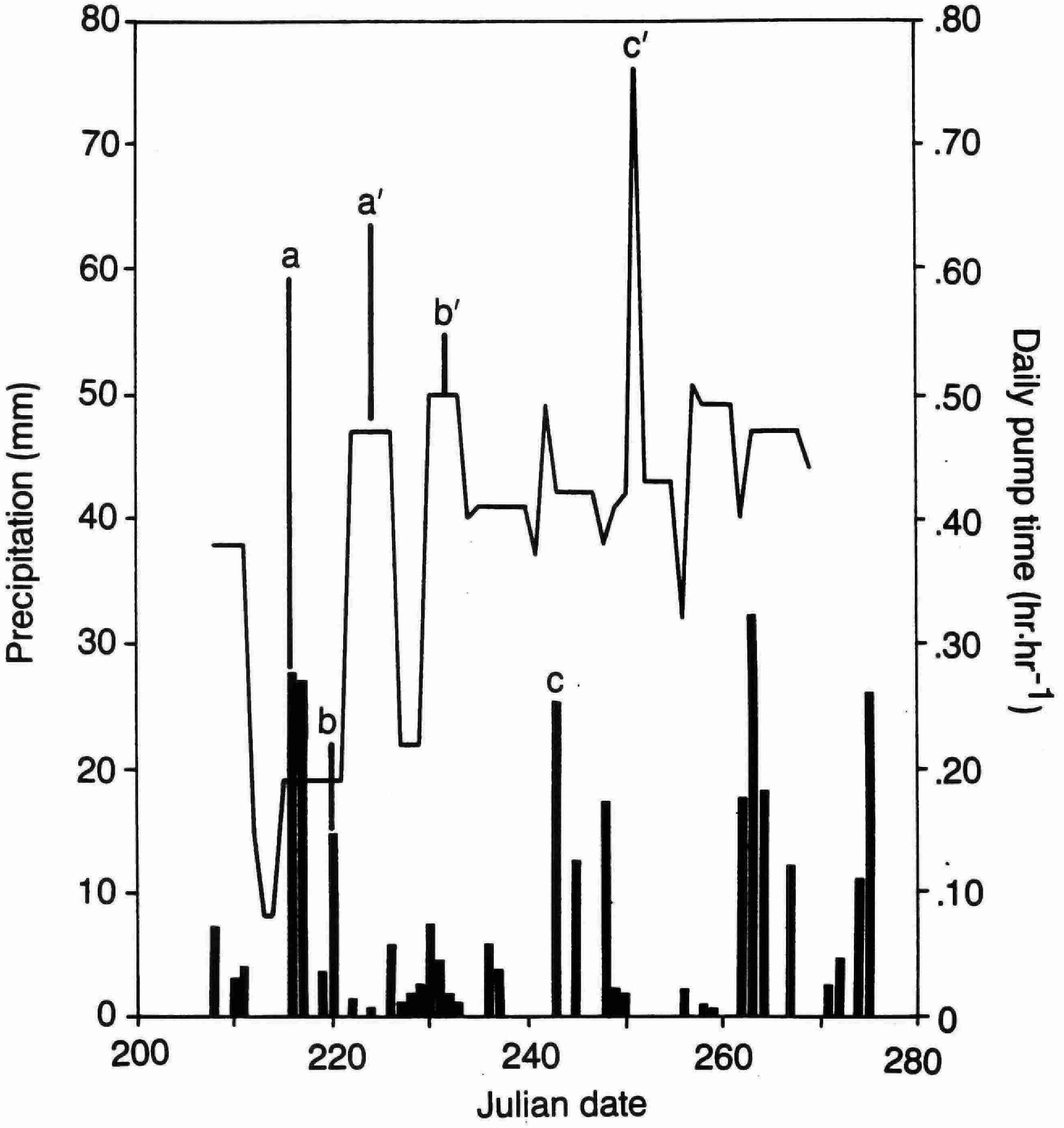


Figure 6.1. Lag time between major rainfall events and leachate peakflow into collection trench.

intercept all the leachate generated, then these data would affirm the postulate that most of the leachate at this site originates from unacceptably high rates of net infiltration of precipitation through the landfill itself. There is thus no significant contribution to leachate generation from groundwater originating from springs beneath the landfill or migrating through the wastes from points offsite.

6.3 Chemical Analysis of Untreated Leachate

As noted in section 3.4, the concentration of various leachate constituents varies as the wastes age and the landfill stabilizes. The changes in plant nutrient content and several other chemical indices of water quality are listed in Table 6.2 for the period from 1978 to 1986. Additional information on inorganic substances in the 1986 leachate can be found in Table 6.8. The variable nature of leachate strength at this landfill is illustrated most graphically by the fluctuating BOD and COD levels. Up to the time of landfill decommissioning (1978-80), a wide range of oxygen demand values were measured. Since 1983, these values appear to have stabilized somewhat in their magnitude and their ratio (i.e. BOD:COD ratio of 0.6 to 1.0). Samples taken from the culvert well recently installed into the contaminated aquifer, however, clearly show BOD and COD values in the anaerobic, groundwater flow environment to be much higher than any previously measured at any other sampling location. This is true for most other constituents as well, particularly hardness and conductivity, which show declines in concentration from the culvert well to the settling lagoon.

It is expected that further biological decomposition of the leachate will decrease the BOD concentrations, thereby further decreasing the current BOD:COD ratio. Tittlebaum (1982) noted that BOD:COD ratios between 0.6 and 0.8 are consistent with MSW leachates and indicate that the wastes are highly susceptible to biological treatment.

Since 1980, both chloride and iron levels appear to have stabilized at the lower end of the concentration range measured in 1978-80. Even with the limited degree of aeration pretreatment achieved in the settling lagoons, sufficient iron is oxidized and precipitated from solution in the lagoons to decrease the concentration from 49 ppm at the sump pump to 11 ppm.

The loss of volatile organics from the culvert well to the spray nozzles is illustrated in Table 6.3. Toluene concentrations measured in 1986 dropped from 1900 to 138 ppb, total xylenes from 100 to 6 ppb and

Table 6.2. Muskoka Lakes untreated landfill leachate constituents: Standard water quality indices (1978-86).

Leachate Constituents	Range over 1978-1980*	Sept. 1983**		July 1986***			
		Sump	Lagoon	Culvert Well	Collection Trench	Lagoon	Discharge to Lake
Plant Macronutrients							
N-total Kjeldahl	3-81	33	33	92	39	33	3
$N-NH_4$	-	30	31	91	37	33	2
$N-NO_3$	-	<0.2	0.2	0.75	0.3	<0.2	<0.15
Nitrite	-	-	-	0.55	<0.03	<0.03	<0.03
Phosphate	-	-	-	0.8	0.2	0.2	0.2
Phosphorus	-	0.2	0.2	0.4	<0.05	<0.05	<0.02
Plant Micronutrients							
Cl	9-147	31	31	83	34	33	7
Fe	11-654	49	11	n.a.	n.a.	n.a.	n.a.
Other Indices							
pH	-	6.05	6.28	6.17	5.76	6.02	7.29
Hardness (as $CaCO_3$)	45-583	295	278	18125	327	301.5	73.5
Alkalinity (as $CaCO_3$)	33-988	350	26	1172	354	312	61
Conductivity	-	1071	868	3440	1122	1009	218
BOD	101-2350	900	1150	3920	1090	892	68
COD	159-4033	-	1180	5170	1180	1600	108
TOC (total organic carbon)	-	-	-	1580	-	-	-
DOC (dissolved organic C)	-	370	378	1660	434	365	31
DIC (dissolved inorganic C)	-	31	30	12	3.2	2.4	7.8
TDS (total dissolved solids)	-	858	732	-	-	-	-
DS (dissolved solids)	-	653	670	-	-	-	-

Units ppm ($mg \cdot \ell^{-1}$) for all but pH and conductivity ($\mu mhos \cdot cm^{-1}$)

n.a. - not available
\- - not tested

* - Totten, Sims, Hubicki Assoc., (1983)
** - Ont. Ministry of the Environment, (1983)
*** - Ont. Ministry of the Environment, (1986)

Table 6.3. Muskoka Lakes untreated landfill leachate constituents: Volatile organics (1983-86).

Leachate Constituents	April 1983* Collection Trench	Leaching Trench	July 1986* Culvert Well	Collection Trench	Lagoon	Spray Nozzle	July 1986** Culvert Well
	$\mu g \cdot \ell^{-1}$						
toluene	1229	854	1900	2000	106	138	560
dichloromethane	350	270	-	-	-	-	-
methylene chloride	-	-	-	-	-	-	100
total xylenes	-	-	100	42	0	6	6
m-xylene	32	21	-	-	-	-	-
o-xylene	17	14	62	25	0	0	-
1,1 - dichloroethane	-	-	23	0	0	0	<1.0
ethylbenzene	17	11	54	23	0	0	15
benzene	9	6	17	8	0	0	4.2
trichloroethylene	6	4	18	0	0	0	13
tetrachloroethylene	-	-	0	0	0	0	2.5
chloroform	2	2	3	0	0	0	2
total organic carbon	-	-	-	-	-	-	1583 $mg \cdot \ell^{-1}$
total organic halogens	-	-	-	-	-	-	13.9 $\mu g \cdot \ell^{-1}$
total phenolics	-	-	-	-	-	-	0.136 $mg \cdot \ell^{-1}$

* - Ont. Ministry of the Environment (1983; 1986)
** - Zenon Env. Inc.
\- - not tested

Also tested in July, 1986 but not detected: bromodichloroethane, bromoform, carbon tetrachloride, chlorobenzene, chlorodibromomethane, 2-chloroethylvinylether, chloromethane, dichlorobenzene, dichlorobromethane, dichloroethane, dichloroethylene, dichloropropylene, dichloropropane, ethylene dibromide, tetrachloroethane, trichloroethane, trifluorchlorotoluene.

ethylbenzene from 54 to 0 ppb. Several volatile organic compounds were also detected in low concentrations in the culvert well but were either absent or below detectable limits in the lagoon and spray nozzle leachate (i.e. benzene, chloroform, dichloroethane, tetrachloroethylene and trichloroethylene). A list of compounds which were tested for but which were not detected is included in Table 6.3.

6.4 Soil Capacity for Attenuation of Leachate in Situ

6.4.1 Station Monitoring Equipment

The 4.3 ha spray area can generally be segregated into three spray regimes based on the leachate application rates observed in 1986 and the past spray history at the site. Figure 6.2 portrays the 24 spray nozzle positions for each of spray areas 1, 2 and 3. A large proportion of the current spray areas 1 and 2 have been irrigated by a succession of 1, 6 and 72 nozzle systems from 1980 to 1986 (i.e. "heavy" spray area). Furthermore, spray area 1 in its entirety received most of the May-June 1986 application due to system malfunctions in the other two spray areas. Spray area 3 and the majority of spray area 2 have been irrigated only since July 1985 via the current 72 nozzle system. Despite the irregularity of the system's breakdown, it is believed that spray area 2 ("moderate" spray area) received comparatively more total leachate during the 1986 spray season than spray area 3 ("light" spray area).

The station monitoring equipment, their respective purposes and locations are displayed in Table 6.4. Station 1 is the "control" station located in an unsprayed area north of the pumphouse and south of the light spray area (Figure 6.2). Station 2 is located in the light spray area and Station 3 in the portion of spray area 2 which was heavily irrigated prior to 1985. A suitable location could not be found within spray area 1 for a full complement of station monitoring equipment due to the shallow depth to bedrock.

The monitoring equipment included an IRAMS Soil Moisture Analyser, a Troxler neutron thermalization depth probe, tensiometers, ceramic-tipped pressure/vacuum lysimeters, piezometers and wedge and AES standard rain gauges. The IRAMS instrument measures volumetric soil moisture content (θ_v) at 0-15 cm below the surface using a time domain reflectrometry technique. The IRAMS measurement of θ_v, coupled with those of the Troxler depth probe (i.e. effective depth 23^+ cm), provide a system for non-destructive and instantaneous monitoring of soil moisture profiles in an irrigated

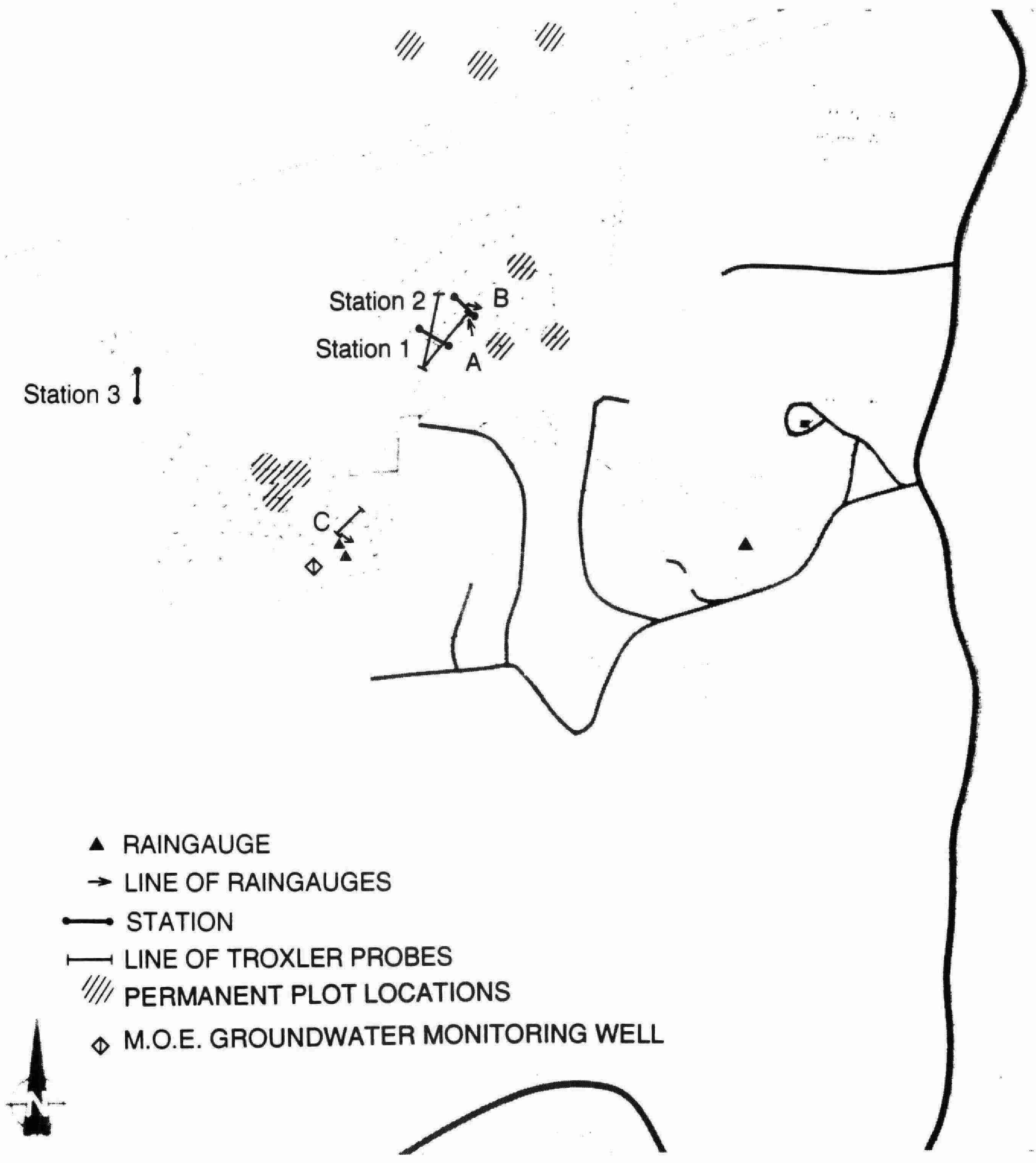


Figure 6.2. Location of field monitoring equipment at the Muskoka Lakes landfill site (scale 1:5,900).

Table 6.4. Soil monitoring equipment, purpose and location.

Soil Monitoring Equipment	Purpose	Station 1 Control (No spray)	Station 2 Light Spray (Spray area 3)	Station 3 Heavy Spray (Spray area 2)	Heavy Spray (Spray area 1)
I.R.A.M.S. Soil Moisture Analyser	Soil moisture 0-15 cm below surface	•	•	•	
Troxler neutron thermalization probe	Soil moisture 23-150 cm below surface	•	•	•	•
Ceramic-tipped tensiometer	Soil Water Pressure Potential at 15, 30, 60, 90, 120 cm below surface	•	•	•	
Ceramic-tipped pressure/ vacuum lysimeter	Collect soil solution at 15, 30, 60, 90, 120 cm below surface	•	•	•	
Standard rain gauge	Depth of spray and/or rainfall	• (one on landfill, one in unsprayed area outside of spray area 1)	•		•
6-7 rain gauges on line from nozzle	Determine spray distribution 0, 2, 4, 6, 8, 10 m from nozzle		• (nozzle A,B)		• (nozzle C)
Piezometer	Position of Groundwater Table	•	•		

environment. A Soil Measurement Systems tensimeter was used in conjunction with nests of variable depth tensiometers to ascertain the negative pressure potential (ψ) profile corresponding to the aforementioned moisture contents. Both the quantity and energy status of water are required to fully characterize the soil moisture regime. Piezometers and pressure/vacuum lysimeters were installed in order to extract groundwater and vadose water samples, respectively, for chemical analysis. AES standard rain gauges were installed on the landfill surface and in an unsprayed portion of the forest to ascertain the quantity of precipitation actually incident on the forest floor. A third standard rain gauge was placed near the non-spray rain gauge in the forest but at a distance of 5 m from a nozzle in the heavy spray area (Figure 6.2). This provided a continuous record beginning in May, 1986 of the spray program in terms of volumes reaching the forest.

6.4.2 Precipitation-Irrigation Regimes in 1986

Data presented in section 4.0 suggested that the Muskoka district experienced an abnormally wet field season in 1986. Figure 6.3 shows that the precipitation conditions at the landfill site (i.e. landfill cap rain gauge) were somewhat drier than at the Muskoka Airport near Bracebridge. This plot also depicts the influence that the tree canopy can have on the seasonal quantity of precipitation that reaches the forest floor. The unsprayed wooded area apparently received about 170 mm less precipitation at ground level from mid-May to the end of September than the landfill area due to evaporation of rainfall from the wetted tree canopy surfaces. Even at a distance of 5 m from a nozzle in the heavy spray area (see section 6.4.3), the spray rain gauge recorded over 200 mm of leachate irrigation during this four-month period, after correction for seasonal precipitation amounts (Figure 6.3).

Figure 6.4 allows for a more direct comparison of the 1986 field season relative to the 30-year climatic norms already presented in Figure 4.4. The precipitation and temperature data required for this construct originated from the Muskoka Airport while the nearest station with daily sunshine hour data was at Honey Harbour. A modified energy budget approach was used in this instance to estimate the daily potential evapotranspiration (Ep), which requires the hours of bright sunshine (or direct net radiation measurements) and daily maximum and minimum air temperatures. The empirical equations used for Ep estimation are given in Appendix C. Clearly, the Muskoka district was in a surplus moisture condition for most of 1986, with only

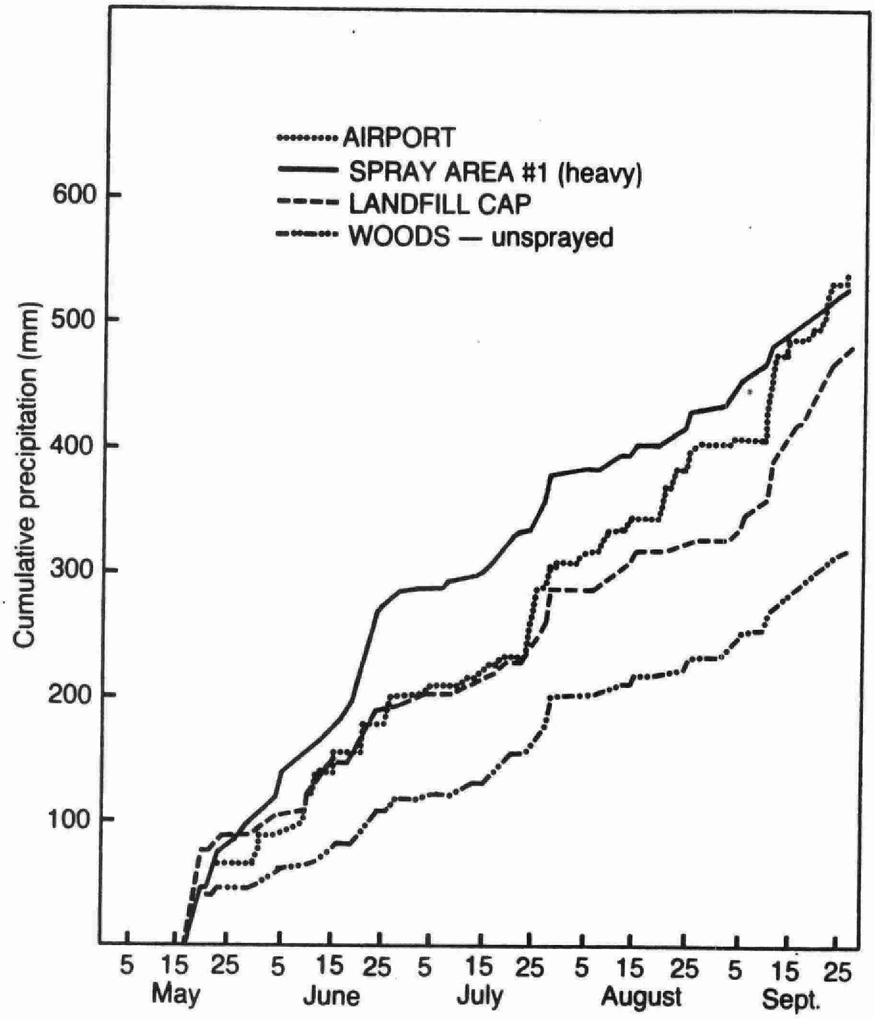


Figure 6.3. Cumulative precipitation (May 18-Sept. 26, 1986) recorded by A.E.S. standard rain gauges at Muskoka Airport and three locations in the landfill vicinity.

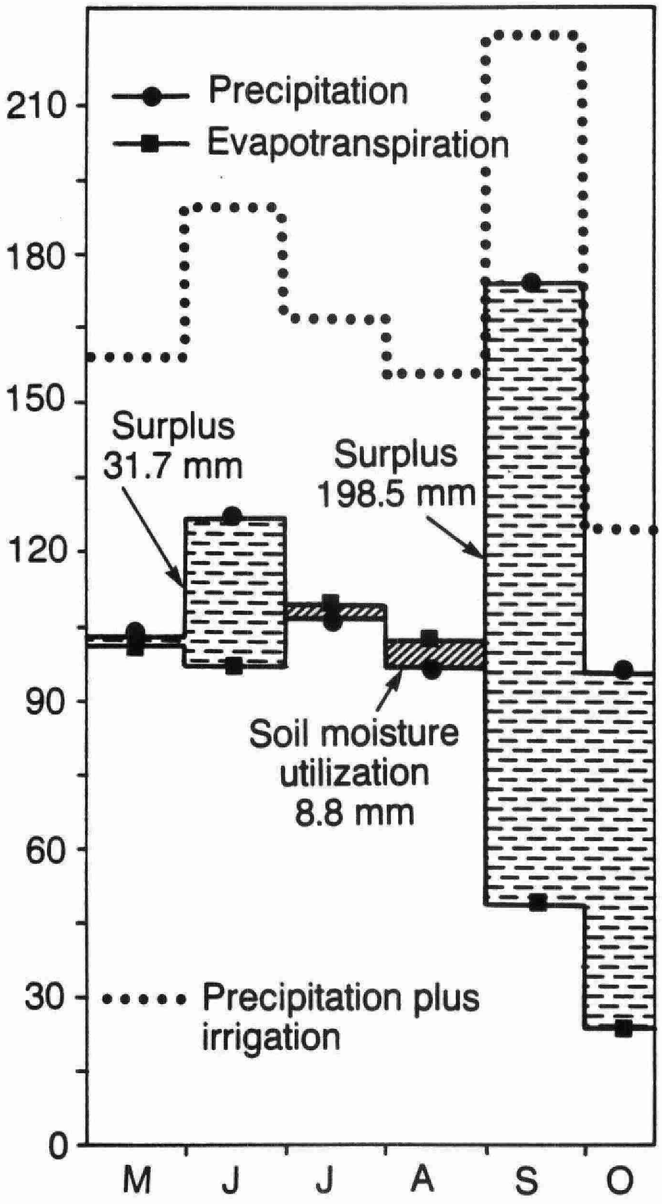


Figure 6.4. Monthly moisture budget based on 1986 precipitation, temperature and sunshine hour data.

very minor quantities of stored soil moisture being required in July and August. A conservative irrigation schedule scenario (i.e. 4 mm of irrigation on each day with no rainfall) was superimposed on Figure 6.4 to underscore the degree of soil waterlogging that was observed on site in 1986.

The minor dry spells that did occur in 1986 can be better isolated by a daily soil moisture budget as seen in Figure 6.5. Soil moisture storage was somewhat depleted by vegetative evapotranspiration in mid-May, mid-July, late August and early September. It is generally acknowledged that little if any effect on plant growth or ultimate productivity will occur until the soil has been depleted of 50% or more of its plant-available storage capacity. Only the mid-July dry spell reaches this point but this deficit condition was very short-lived and likely had little effect. Superimposing an irrigation schedule of the type proposed above would eliminate all evidence of stored soil moisture utilization from Figure 6.5.

6.4.3 Spray Distribution

The current 72 nozzle leachate distribution network employs Rain Bird Model No. 30 nozzles with a 4.4 mm aperture and a capacity and spray radius of 22.8 $\ell \cdot min^{-1}$ and 13.7 m, respectively, at about 250 kPa pressure. To ascertain the distribution of spray over the landscape with distance from individual nozzles and with distance between nozzle positions and the pumping source, lines of up to 7 wedge rain gauges were established in the light and heavy spray areas at 2 m intervals from 3 selected nozzles (Figure 6.2; Table 6.3).

Total accumulations, minus any incident precipitation, were used to examine spray distribution versus distance from the spray nozzle (Figures 6.6a, b and c). An index of distance from the pumping source was calculated as the ratio of distance from the pumphouse to nozzles B or C relative to the distance from the pumphouse to nozzle A. The wide range of spray volumes collected is a result of recording the accumulation of one or more spray cycles on four different trial dates.

Nozzles A and B are on laterals connected to the same main line and were, therefore, operating simultaneously under comparable wind conditions and for an identical duration of time. They clearly did not deliver the same total volumes of spray, however, nor were the spray patterns identical (Figures 6.6a and b). Nozzle B is located 3.2 times farther from the pumphouse than nozzle A. The head loss or pressure difference according to hydraulic theory effectively accounts for the different volumes delivered,

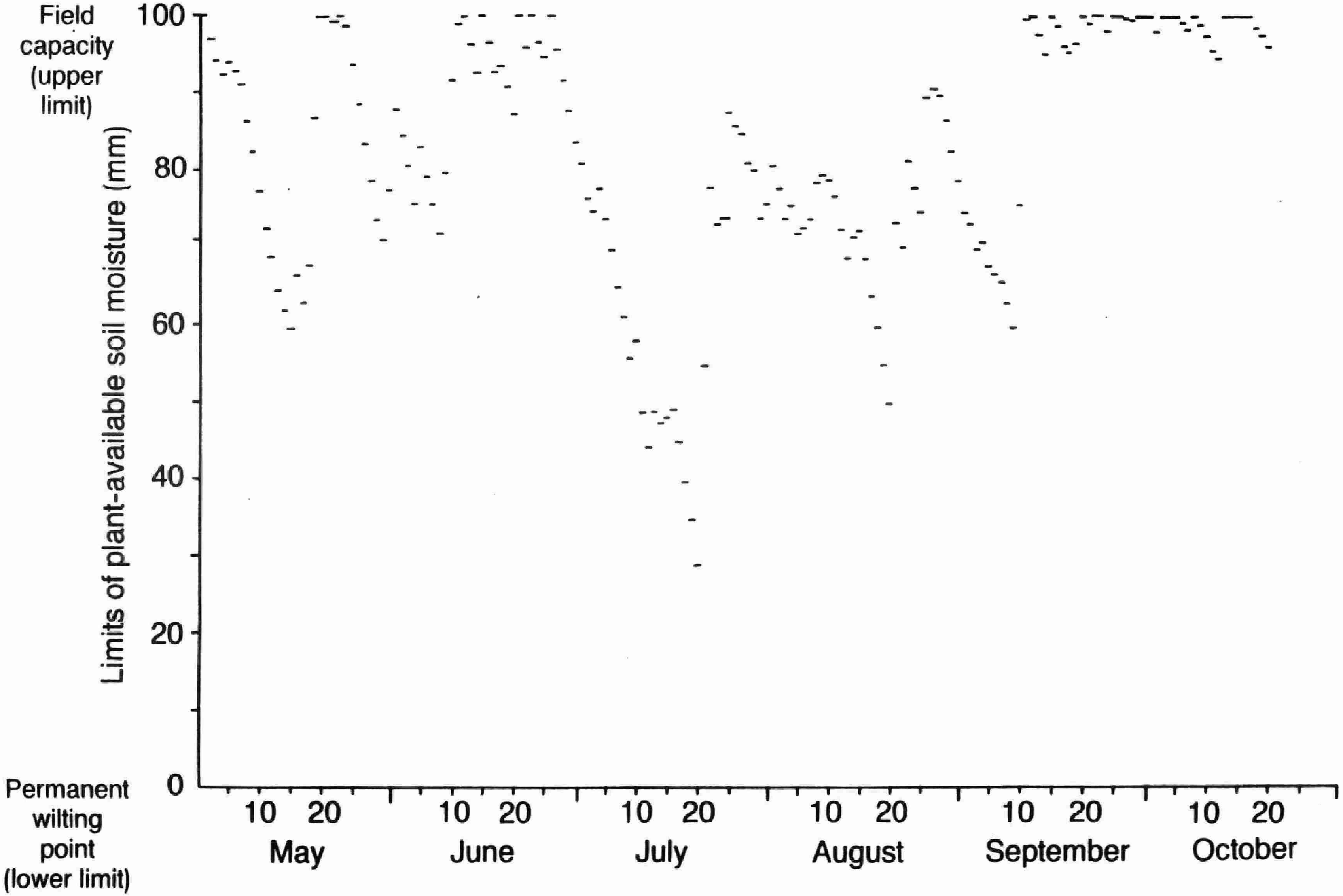


Figure 6.5. Daily estimates of soil moisture status based on an available soil moisture-holding capacity of 100 mm and 1986 precipitation, temperature and sunshine hour data.

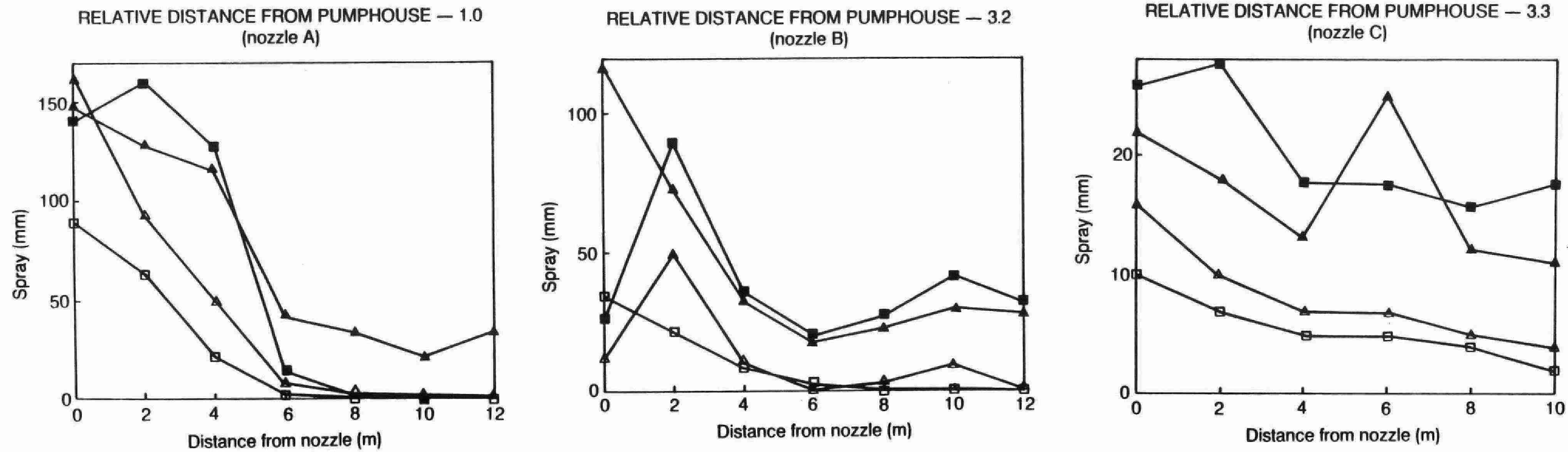


Figure 6.6. Spray distribution pattern a) from nozzle A, b) from nozzle B, c) from nozzle C, with relative distances of 1.0, 3.2 and 3.3 from the pumping source.

particularly at the 2 and 4 m distances. In general, the observed spray distribution pattern was an exponential decrease in the volume of spray delivered 2-6 m from the nozzles. As the total quantity of leachate applied increased, however, the 2-6 m distribution pattern began to appear more parabolic. Beyond 6 m, the distribution tended to level off or drop to negligible volumes due in part to the condition and maintenance of the Rain Bird nozzles themselves.

The leachate volumes immediately adjacent to the nozzle (i.e. 0 m) are quite erratic and should be viewed with caution. It was frequently noted that back pressure at some nozzle positions caused leachate to flow as an unatomised stream from the nozzle to a distance of up to 1 to 1.5 m from the standpipe. Since the resting position of the pivoting nozzle varied after each spray cycle, the rain gauge at 0 m collected extremely variable quantities of leachate compared to the rain gauges at 2 m and beyond (Figure 6.6b). For this reason, extremely wet soil conditions were commonly observed at the base of most nozzles throughout the spray season.

Nozzles B and C (Figures 6.6b and c) are both located roughly 3 times farther from the pumphouse than nozzle A. Despite comparable trial spray durations, the lesser volumes of spray collected at nozzle C would seem to have resulted from iron and particulate organic deposits clogging the standpipes and nozzles. This further reduces the pressure at nozzle C and gives rise to a less characteristic but more evenly distributed spray pattern.

In the forested spray area at Muskoka Lakes, therefore, it would seem that the more common sources of spray distribution variability such as wind conditions are superseded by i) the considerable magnitude of pressure losses along the three widely-spaced networks of 24, non-overlapping spray plots, and ii) the high long-term irrigation system maintenance requirements.

6.4.4 Physical, Chemical and Morphological Soil Characterization

A total of six soil pits were excavated on site for the purpose of full pedological characterization. These data are assembled in four tables (D.1 - D.4) in Appendix D. Soil pits 1-4 represented a catena or toposequence in the deep sands of spray area 3 (light spray) which began at an imperfectly drained position near a depressional area (pit 1) and terminated in an excessively well drained landscape position (pit 4). This soil catena followed a linear transect such that pits 1 and 2 lay outside of the spray

area (i.e. pit 2 was situated at station 1 (see Figure 6.2)). Pits 3 and 4, however, were within the spray area (i.e. pit 4 was situated at station 2) but at different distances from a fixed spray nozzle position (Table D.1). Pit 5 was excavated near station 3 (Figure 6.2) in the heavy spray area and pit 6 was within one of the permanent 0.05 ha vegetation plots (section 7.1) in the control area on the north side of the hydro transmission corridor.

Table D.1 provides much of the more descriptive information typically recorded in a full pedological characterization as well as the leachate spray regime at each pit location. The unsprayed soils (pits 1, 2, 6) belong to the Sombric or Dystric Brunisol Great Groups (C.S.S.C., 1978), depending on whether the LFH/Ah horizon is greater or less than 10 cm in depth. The sprayed soils (pits 3, 4, 5), however, generally belong to the Ferro-Humic Podzol Great Group due to the presence of very well developed podzolic B horizonation. The elevated levels of sesquioxides and organometallic complexes in these podzolic B horizons are undoubtedly a direct result of the excessive loading of iron-laden wastewaters on these soils which were in a natural embryonic stage of podzolization prior to the spraying activity. Of particular note are the iron-indurated or cemented Bfc and Bhfc horizons which were found to be quite continuous within the spray areas but were better developed nearer the nozzle positions, due presumably to heavier leachate loading rates (section 6.4.3). These placic layers ("ortsteins") can greatly impede the movement of soil water, causing perched water tables within the root zone, and obstruct the development of root systems due to high mechanical impedence. Cementation was also noted more sporadically in the C horizon at some locations (e.g. pit 2) due to the combined effects of SiO_2, Fe_2O_3 and Al_2O_3 precipitation (DeKimpe et al., 1983) and the formation of fragipan/duripan layers. It is not known at this time if the spray operations are directly responsible for this subsoil cementation but the elevation of pH in the sprayed soils (Table D.4) may provide an environment more conducive to silica and sesquioxide precipitation.

Tables D.2 and D.3 present the more pertinent physical properties of these soils including particle size distribution and soil moisture retention characteristics. The sand contents range from about 66% to over 99%, placing most soil horizons in the sand to loamy sand textural classes. The dry bulk densities are quite low within the upper root zone of the forest soils but increase to more typical values for these textures at depth (i.e.

1.5 - 1.6 $g \cdot cm^{-3}$). There is a marked tendency for the fully or partially developed podzolic B horizons in these soils to have lower dry bulk densities than horizons immediately above or below. This is thought to be due to the higher measured organic matter contents and the presence of illuviated organometallic complexes. Despite the lower densities, these horizons are virtually impermeable to water migration or root ramification. Figure 6.7 depicts the soil moisture retention curves for three horizons (pit 1) as taken from Table D.3. This plot exemplifies the importance of the forest floor litter layer in leachate retention above the groundwater table. The LFH/Ah horizon would be in a constant state of flux with irrigation between a near saturated state and the permanent wilting point (i.e. θ_v range of 86% to 39% $m^3 \cdot m^{-3}$) whereas the Cgj horizon would remain closer to a metastable equilibrium at field capacity (i.e. θ_v of 29% to 17% $m^3 \cdot m^{-3}$). In an unirrigated condition, these soils are subject to appreciable droughtiness due to deep groundwater tables and a low plant-available water-holding capacity (θ_{aw}). Table D.3 shows that θ_{aw} is generally in the 10% $m^3 \cdot m^{-3}$ range in the solum (topsoil) mineral horizons but drops dramatically in the subsoil to as low as 2% $m^3 \cdot m^{-3}$. In terms of potential forest (fibre) productivity, these soils would most likely belong to Class 3 according to the Canada Land Inventory System for Forest Capability and would become submarginal where bedrock outcrops predominate.

The principal chemical properties of the six soil profiles are given in Table D.4. Of note are the very low cation exchange capacities, particularly in the sandy subsoils where secondary clay minerals and organic matter are virtually absent. The highest C.E.C. values are in the LFH/Ah horizon due to the pH-dependent exchange sites in the accumulated surface organic matter. This again underscores the importance of the forest floor litter layer in terms of one of the principal wastewater renovation processes, namely cation exchange. In an unsprayed environment, pH tends to increase in the soil profile with depth, with the low pH in the surface horizons caused primarily by the formation of organic acids from organic matter decomposition and the leaching of basic cations. Pit 5 in the heavy spray area, however, shows that longer term leachate applications tend to elevate the pH in these surface horizons due to the soil's low buffering capacity. This provides an environment condusive to sesquioxide precipitation in the B horizon and the formation of ortstein features. Nitrate, Ca, K and Fe levels appear to increase most significantly in response to prolonged leachate applications.

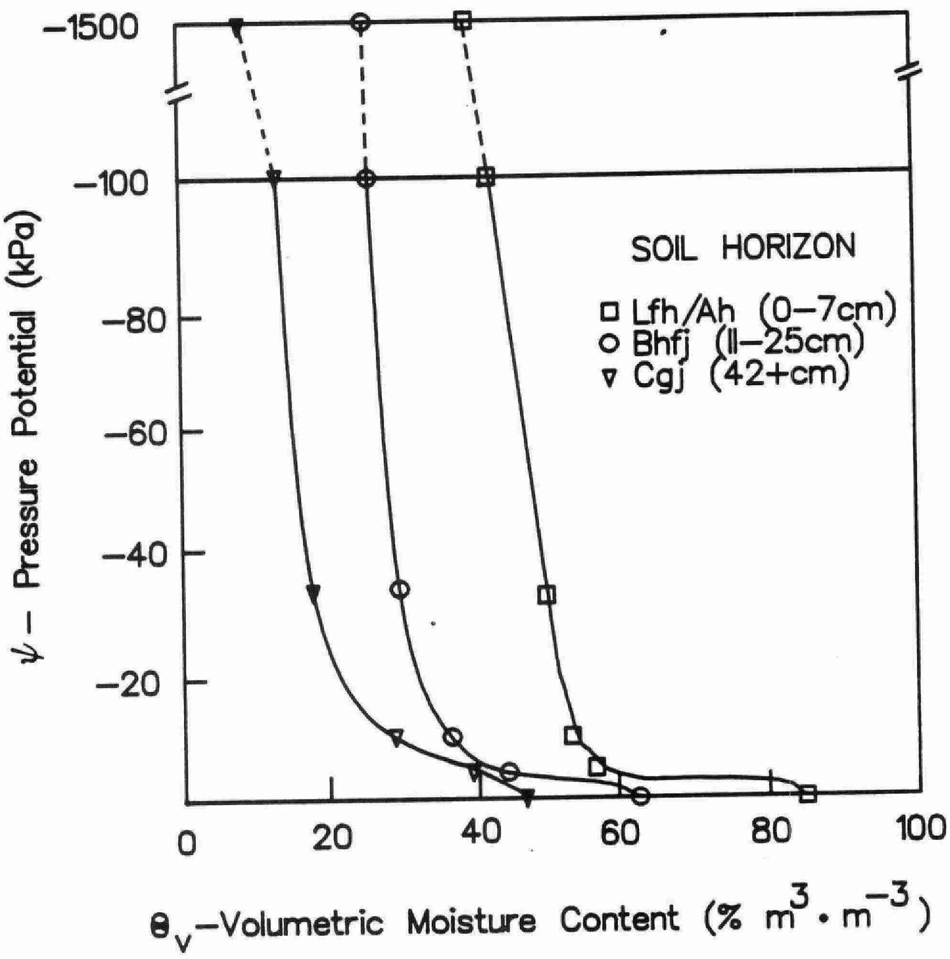


Figure 6.7. Measured $\psi(\theta)$ soil moisture retention characteristics for three soil horizons from an unsprayed soil (pit 1).

6.4.5 Soil Water Regimes Under Sprayed and Unsprayed Conditions

Periodic assessments of the soil water regime at stations 1-3 were carried out to gain a better understanding of the prevailing soil aeration/plant-available moisture status of the local soils under both irrigated and non-irrigated conditions. To illustrate these moisture content (θ_v) - negative pressure potential (ψ) relationships, data from two dates are presented in three figures in Appendix E (Figures E.1, E.2 and E.3). On August 7, 2.6 mm of rainfall were recorded and no spraying was undertaken. On September 17, no rainfall was recorded but a heavier than normal perturbation spray schedule was implemented involving three full 1-hour cycles (0.5 hour interval between alternating cycles) in all spray areas. The soil moisture regime was monitored 2 hours after the last cycle was complete to allow for leachate redistribution in the soil profile.

From August 7 to September 17, the unsprayed area at station 1 (Figure E.1) shows that soil conditions were somewhat wetter in the surface 100 cm due presumably to the well above average precipitation in September. The measured pressure potential on August 7 shows a clear wetting front at the 30 cm level as a result of the recent rainfall event, but it was not heavy enough to cause a marked change in the θ_v measured at that level. Overall, the θ_v tends to decline with depth and falls to very low values in the sandy subsoil due to the deep groundwater table position.

Station 2 (Figure E.2) is situated on a higher landscape position with a deeper groundwater table, although heavy rainfalls in September caused the phreatic surface to rise from greater than 400 cm in early August to 332 cm below the surface on September 17. Figure E.2a shows the same wetting front as in Figure E.1a arising from a recent rainfall but the θ_v is generally much lower than at station 1 in the surface 140 cm. The post-spray moisture profile (Figure E.2b) on September 17 shows a relatively consistent increase in θ_v of about 3-5% $m^3 \cdot m^{-3}$ throughout the full root zone as a result of the heavy irrigation earlier in the day and past rainfall. The ψ values show a characteristic steady decline with depth but will tend to increase toward 0 kPa in the capillary fringe above the groundwater table at 332 cm depth.

Station 3 (Figure E.3) is clearly the wettest site monitored as evidenced by much increased θ_v and ψ values relative to stations 1 and 2. The regional groundwater table tends to correspond to the depth to bedrock on this site (i.e. about 150 cm deep), although frequent perched water

tables were observed due to orstein layer formation in these soils. Spray irrigation clearly exacerbates an already serious excess water problem (Figure E.3a) by elevating both θ_v and ψ to levels where soil aeration will become limiting to vegetative growth and influence soil microbial activity (Figure E.3b). The rapid increase in θ_v with depth shows that the moisture content profile is intimately linked to the position of the groundwater table. Ponding of excess leachate and precipitation on consolidated bedrock can induce marked and rapid phreatic surface fluctuations. Unstable soil moisture regimes of this type greatly limit the diversity of vegetation types which can survive under these transient conditions.

6.4.6 Subsurface Water Analyses in Spray Areas

6.4.6.1 Soil Solution Quality

Despite the low water retention capacity of the local soils, attempts were made to extract soil solution as a field assessment of soil attenuation capacity given that the soil moisture contents would be maintained artificially high from leachate irrigation. There is little agreement in the literature, however, about the validity of soil solution samples extracted with porous ceramic cups in a pressure/vacuum lysimeter apparatus. Grover and Lamborn (1970) found that the ceramic material contributed excessive amounts of Ca, Na and K to extracted solutions and adsorbed significant amounts of P. Hansen and Harris (1975) found that solution P content varied by up to 60% due to factors such as sorption, leaching, diffusion and screening by the ceramic material. Nitrate was similarly affected by sampler intake rate, plugging of pores, depth of lysimeter placement and type of vacuum system utilized. Conversely, Levin and Jackson (1977) have suggested that Ca, Mg and P levels are not affected by ceramic cup lysimeters but NO_3^- appears to be screened out. Recognizing these potential shortcomings of pressure/vacuum lysimeters, this technique was used at the Muskoka Lakes site since it is the only proven field method yet developed for soil solution extraction. It is also much more expeditious for routine sampling than laboratory-based procedures involving immiscible displacement.

Three nests of ceramic-tipped pressure/vacuum lysimeters were placed at 15, 30, 60, 90 and 120 cm depths below the surface at monitoring stations 1 (control), 2 (light spray) and 3 (heavy spray). Soil solutions were collected once per week during August and September a short time after the

daily leachate spray cycle was completed. The solutions were analysed for pH and several macro- and micro-nutrients as listed in Table 6.5. Nitrates were not analysed for due to the acknowledged screening effect mentioned above. No attempt was made to obtain volatile organic concentrations due to the difficulty of controlling volatilizaton into the headspace above the solution in the lysimeter. The mean concentrations of these constituents found in solutions extracted from 15 and 90 cm below the surface are presented in Table 6.5. Disequilibrium in the soil solution is evident from the very high pH values measured and is believed to be caused by exposure to the ceramic material.

Few definitive trends with soil depth or past spray history are evident from these data. Ammonium and P were detected at less than 1.0 mg•ℓ^{-1} at 15 and 90 cm depths at all three stations. Concentrations of NH_4^+ tended to increase with higher leachate applications in the surface soil, due possibly to inhibition of nitrification of mineralized N in more heavily irrigated regimes. Only the heavy spray area showed detectable levels of NH_4^+ at 90 cm. The confounding effects of P sorption and K, Ca and Mg contamination

Table 6.5. Mean concentrations (mg•ℓ^{-1}) of several soil solution constitutents extracted from two depths (August - September, 1986).

Spray Treatment	Soil Solution Analyses							
	NH_4^+	P	K	Ca	Mg	Mn	Fe	pH
	mg•ℓ^{-1}							
				15 cm Below Surface				
Control	0.13	0.61	16.36	41.76	68.34	0	0	8.55
Light	0.40	0.66	30.80	62.55	49.70	0.07	0	8.40
Heavy	0.73	0.56	21.75	48.68	46.33	0	0	8.61
				90 cm Below Surface				
Control	0	0.37	31.20	63.30	35.80	0	0	9.00
Light	0	0.62	29.23	81.12	83.68	0	0	8.35
Heavy	0.25	0.62	30.97	64.55	54.45	0.03	0	8.48

caused by the ceramic porous material appear to obscure any significant trends in the concentration of these nutrient elements with either soil depth or level of spray treatment. Under conditions of more extreme perturbation spray trials, however, available P tended to increase with the spray rate. For example, in the heavily sprayed area, P levels of 0.55 and 0.78 mg•ℓ^{-1} were detected in solution samples taken from a 15 cm depth after

intentionally heavy daily applications of 6 and 17 mm, respectively. The P concentration was 0.35 $mg \cdot \ell^{-1}$ at 15 cm before these spray trials began. Screening or sorption of Fe and Mn by the ceramic material was not ruled out in explaining the virtually undetectable levels of these metals found in the extracted solution.

A single teflon-tipped pressure/vacuum lysimeter was installed as part of the lysimeter nest at station 2 (light spray) in an attempt to quantify the extent of the sorption-contamination effects of the ceramic material on soil solution chemistry. Despite repeated efforts, it was not possible to extract the soil solution due to the failure of the inert teflon tip to wet properly and avoid air bubbling under the low moisture content regime of the local soils.

6.4.6.2 Groundwater Quality

Groundwater monitoring wells were established at three locations. The first was situated at station 2 in the light spray area (see Figure 6.2). The second was situated outside of the spray area at the southern extremity of the same catena about 3 m from a natural depression where leachate break-outs frequently occur. Analysis of water samples obtained from these wells will permit regular "effects monitoring" on the degree of groundwater contamination attributable to slow rate soil infiltration disposal of leachate. This is also true for the third groundwater monitoring well which was installed prior to 1986 and this study near the southern boundary of spray area 1. In addition, the piezometer network (Figure 5.4) and the culvert well provide many additional points where groundwater samples can be obtained to ascertain the degree and possible expansion of aquifer contamination in the vicinity of the landfill itself.

Chemical analyses of water samples taken from each of the three groundwater monitoring wells on the west side of the perennial stream are presented in Table 6.6. These data indicate higher groundwater contaminant concentrations in the vicinity of spray area 1 (heavy spray) when contrasted to those obtained in samples from spray area 3 (light spray). Concentration levels in samples obtained from the break-out area were also greater than those found a short distance upgradient at station 2 despite the fact that the former location is not within a spray area. These observations are to be expected if consideration is given to both the spray irrigation history

Table 6.6. Chemical analysis of groundwater samples (October 22, 1986).

Location	Water Analysis ($mg \cdot \ell^{-1}$)					
	NO_3^-	P	K	Ca	Mg	Fe
Station 2	0	.05	0	2.9	0.7	0
Break-out depression	0	.05	4.9	8.8	3.1	0.5
M.O.E. well	0	.05	14.5	11.6	5.0	0.5

of the area and the local physiography. In spray area 1 (M.O.E. well), the overburden is relatively shallow and has been subject to seasonal leachate irrigation since 1980. One or both of these factors has created a comparatively low soil attenuation capacity over time and increased the risk of groundwater contamination. The overburden in the vicinity of spray area 3, however, is comparatively deep and has only been spray irrigated since July 1985. Within the catena or toposequence in spray area 3, the break-out well has been installed with the intent of intercepting groundwater from an intermediate scale flow system where it surfaces in a natural depressional area. It is thought that the water quality in this flow system is affected by incompletely renovated leachate reaching the groundwater table as deep drainage within a micro-watershed with dimensions at least encompassing spray area 3 in its entirety. Samples acquired from the station 2 well are representative of a more local scale flow system and thus contaminant concentrations would be anticipated to be somewhat lower.

Maximum Fe concentrations of 0.5 $mg \cdot \ell^{-1}$ attest to the near complete removal of iron in the unsaturated zone of the overburden (i.e. about 50 mg $Fe \cdot \ell^{-1}$ in untreated leachate). The effect that this iron accretion has on soil morphology has been discussed in section 6.4.3.

Potassium is a relatively mobile cation, particularly under conditions where the local soils are essentially devoid of secondary clay minerals which can fix this element in their lattice structure. As such, K is a good index of the degree of groundwater contamination at the Muskoka Lakes site as indicated in Table 6.6. Calcium and Mg also appear to increase as groundwater quality deteriorates due to leaching from the irrigated soil profiles as these basic cations are displaced by Fe and other cations.

6.4.7 Field Soil Water Conductivity and Infiltration Patterns

Using the recently developed Guelph Permeameter, field saturated hydraulic conductivity (K_{fs}) of the local soils was measured at several locations in situ. The information is necessary to gain a better

understanding of some of the groundwater and vadose water quality results presented in section 6.4.6 above. A mean K_{fs} of 7.5×10^{-3} $cm \cdot s^{-1}$ was obtained at the selected measuring depth of 45 cm in the proximity of spray area 3 where the fine and medium sand overburden is approximately 5-6 m deep. This K_{fs} value corresponds to the high conductivity (H_1) category of the Agriculture Canada soil macrostructure description scheme (McKeague _et al._, 1986). In the course of computing K_{fs} from the field data, it was observed that a significant proportion of the resultant values were negative. Reynolds _et al._ (1983) deduced that negative K_{fs} values indicate the presence of hydrologic discontinuities, typically caused by soil stratification or the presence of biopores (e.g. rodent, earthworm, root holes). It was observed in the course of detailed soil characterization (Table D.2), however, that the soil stratigraphy was relatively homogeneous at 45 cm below the soil surface (i.e. K_{fs} measuring depth). The negative K_{fs} values were thus due to root pores since there was no evidence of burrowing animal activity and acidic soils are not condusive to the establishment of earthworm populations. Knowledge of the effect of these root channels on surface infiltration patterns and subsurface moisture distribution within the overburden was critical to ascertaining appropriate leachate application rates on this site.

An infiltration experiment was undertaken using a methylene blue dye solution to characterize vertical and lateral flow along the soil catena spanning spray area 3. At five locations, soil pits were excavated to a depth of about 2 m. Infiltrometer rings of 30 cm diameter were installed at a distance of 1 m from each pit and were embedded to a depth of about 10 cm to allow proper ponding and infiltration of the dye solution. At four of the pits, 20 ℓ of dye solution (0.1% $g \cdot g^{-1}$) were poured into the infiltrometer rings whereas 80 ℓ were applied at the fifth pit. Following complete infiltration of the dye solution, the appropriate face of each soil pit was cut back incrementally 15 cm at a time. After each new pit face was exposed, vertical and lateral flow patterns were visually observed and recorded.

The results of the experiment indicated that significant quantities of surface applied leachate can move rapidly to depths beyond the root zone by preferential flow (i.e. "short circuiting") along old root channels (Moore _et al._, 1986). The results were very similar regardless of the volume

applied or the spray history of the soil. In each of the 5 replications, it was evident that downward vertical flow far exceeded any lateral or horizontal flow. In fact, the only lateral translocation of dye solution was associated with rapid root channel flow. The volume of soil encompassed by the dye wetting front was smaller than expected due to this preferential flow and the apparent adsorption of the methylene blue (i.e. a large organic molecule) within the forest soil litter layer.

When the results of the saturated hydraulic conductivity and infiltration (dye) investigations are combined, it is apparent that leachate residence time above the groundwater table, and therefore leachate attenuation, is likely to be highly dependent on the ability of the litter layer to retain leachate and diminish infiltration rates.

6.5 Soil Capacity for Attenuation of Leachate in Soil Columns

A major constraint evident from discussions in section 6.4 involving the assessment of soil attenuation capacity for leachate contaminants in a field environment was the lack of control over both past and present spray operations at the Muskoka Lakes site. The purpose of this segment of the study was thus to establish soil leaching columns in a controlled laboratory environment with a view to:

i) ascertaining the attenuation capacity of local soils for untreated, activated carbon-treated and lime-treated leachate

ii) simulating the formation of an iron-indurated layer within the columns

iii) determining if long-term spraying influences attenuation capacity (i.e. both previously sprayed and unsprayed soils were used for the columns)

iv) evaluating the importance of the litter layer on overall soil attenuation capacity (i.e. litter layer truncated from some columns).

Thirty soil columns (4.5 cm x 70 cm) were extracted fully intact within clear acrylic tubes at two locations; 15 from an unsprayed (control) area and 15 from the area which had been sprayed since 1980 (heavy spray treatment). Raw leachate drawn from the collection trench, lime-treated leachate (2.0 g $Ca(OH)_2 \cdot \ell^{-1}$), leachate which had been passed through a column of granular activated carbon, and a control of rainwater collected at the Muskoka site were applied daily to the columns. Rates of application

were 3.5, 7.0 and 14.0 $mm \cdot d^{-1}$ with the daily aliquot being regulated by drip funnels placed above the columns. Four previously sprayed and four unsprayed soil columns were truncated (i.e. litter layer and Ah horizon removed) and subjected only to intermediate application rates (7.0 $mm \cdot d^{-1}$). This would further allow examination of the attenuation capacity of the local soils should a subsurface irrigation disposal system be installed at the site. Effluent which percolated through each soil column was collected weekly and analysed for NO_3^-, NH_4^+, P, K, Ca, Mg, Mn, Fe, and pH. The column treatment permutations were not replicated.

Leaching experiments of this type are often very long-term ventures with up to 10 pore volumes required before an equilibrium condition is reached. Daily applications are thus still ongoing and will continue until all columns exhibit a stabilized attenuation or elution pattern at which point the columns will be destructively analysed to determine the vertical distribution of these chemical constituents. At the time of writing, the columns had received between 0.5 and 3.0 pore volumes of leachate, depending upon the rate of application. Hence, the major trends of attenuation or elution reported here for each leachate constituent are those observed from the highest loading rate columns (i.e. 14.0 $mm \cdot d^{-1}$) since they have provided the most complete pattern to date (Table 6.7). The results for the truncated and intact columns at the intermediate application rate can be found in Appendix F.

In order to compare treatments, the relative concentration (R.C.) of each constituent was calculated as the ratio of the column effluent concentration divided by the influent concentration. An R.C. >1.0 thus indicates elution or displacement of a chemical constituent from the soil exchange sites whereas an R.C. <1.0 indicates that attenuation or adsorption is occurring within the soil profile. Once the experiment has been terminated, the "mean attenuation number" for each chemical constituent will be determined following the integration method of Griffin et al. (1976). Table 6.8 lists the mean pH and influent concentration of constituents delivered to the soil columns.

The predominance of either attenuation or elution processes involving specific cations or anions in solution is largely determined by the cation (C.E.C.) and anion (A.E.C.) exchange capacities of the soil. The C.E.C. is the sum total of exchangeable cations that a soil can adsorb on its

Table 6.7. Relative concentration and attenuation/elution patterns of leachate constituents from intact leaching columns subjected to the high application rate (14.0 mm•d^{-1}).

Constituent	Non-Sprayed				Sprayed (1980-86)			
	Untreated	Lime-Treated	Carbon-Treated	Rainwater (Control)	Untreated	Lime-Treated	Carbon-Treated	Rainwater (Control)
$N\text{-}NH_4^+$	0.0-0.16 A	0.0 A	n.c.	n.c.	0.0 A	0.0 A	n.c.	n.c.
$N\text{-}NO_3^-$	0.9-7.5 E→A	1.7 E	1.4-4.3 E	1.0-21.0 E	1.4-33.0 E	0.5-1.3 E→A	0.7-2.9 E→A	1.3-75.0 E
P	n.c.	n.c.	n.c.	n.c.	n.c.	n.c.	n.c.	n.c.
K	0.03-0.06 A	0.0-0.02 A	0.3-0.8 A	0.08-0.2 A	0.4-0.97 A	0.3-0.7 A	0.002-0.5 A	3.4-12.3 E
Ca	0.04-0.2 A	0.03-0.5 A	0.2-0.8 A	0.07-9.2 E→A	0.2-0.6 A	0.04-1.4 E→A	0.07-0.4 A	0.09-26.8 E→A
Mg	0.06-0.1 A	0.8-2.5 E→A	0.0-1.6 E→A	0.08-6.9 E→A	0.4-2.7 E→A	0.6-25.0 E→A	0.1-1.5 E→A	0.27-45.8 E→A
Mn	0.01-0.08 A	1.0-2.7 E	0.0-3.0 E	0.002-2.5 E→A	0.03-0.25 A	0.33-14.0 E→A	7.0-20.0 E	0.0-8.0 E→A
Fe	0.0-0.1 A	n.c.	n.c.	0.0 A	0.0-0.02 A	n.c.	n.c.	0.0 A

n.c. - relative concentration not calculated since influent concentration below detectable limits
A - attenuation
E - elution
E→A - initial elution, subsequent attenuation
A→E - initial attenuation, subsequent elution

Table 6.8. Mean influent pH and concentration of leachate constituents delivered to soil columns to February, 1987.

Treatment	pH	NO_3^-	P	K	Ca	Mg	Mn	Fe	NH_4^+
		$mg \cdot \ell^{-1}$							
Rainwater	6.30	7.4	<0.1	3.5	2.0	16.5	0.1	1.3	<0.1
Untreated leachate	6.30	5.6	<0.1	37.6	93.1	18.5	2.8	10.7	28.7
Carbon - treated	7.92	12.5	<0.1	49.5	79.0	21.3	0.1	<0.01	<0.1
Lime - treated	8.24	1.6	<0.1	48.7	177.9	8.0	0.7	<0.01	4.2

negatively-charged exchange sites. This property is a function of the charge density on the secondary clay minerals (permanent change) and on the soil organic matter (pH dependent charge). Conversely, the A.E.C. is a measure of the total exchangeable anions that a soil can adsorb and approaches zero at pH > 5.0. The "point of net zero charge" is the pH at which C.E.C. equals A.E.C. For soils of the type found at Muskoka Lakes, this net zero charge will occur at about pH 3.9. Hence, at pH values greater than 3.9, which are characteristic of the Muskoka Lakes soils, cation and not anion exchange will be the predominant soil adsorption process. Furthermore, no anion exchange capability will exist in soils which have been previously sprayed. Table D.4 shows that the upper solum horizons have become increasingly alkaline with leachate application with the LFH/Ah horizon pH increasing from pH 3.7 in the control (soil pit 6) to pH 6.4 in the heavily sprayed area (soil pit 5).

Table 6.7 and Tables F.1 and F.2 (Appendix F) show several instances where a R.C. could not be calculated due to influent concentrations below detectable limits (e.g. P for all leachate treatments). Otherwise, a range of R.C. values has been reported on these tables as well as the predominant elution and/or attenuation pattern observed from graphs constructed similar to those in Figures 6.8 and 6.9.

As expected, anionic NO_3^- formed by the nitrification of mineralized soil N showed strong elution patterns (Figure 6.8) and cations were attenuated to varying degrees, although certain cations showed some elution before the R.C. fell below 1.0 (Figure 6.9). The elution of Mg in particular is caused by its displacement on soil exchange sites by more readily adsorbed cations such as Fe. Only Fe and Mn are strongly

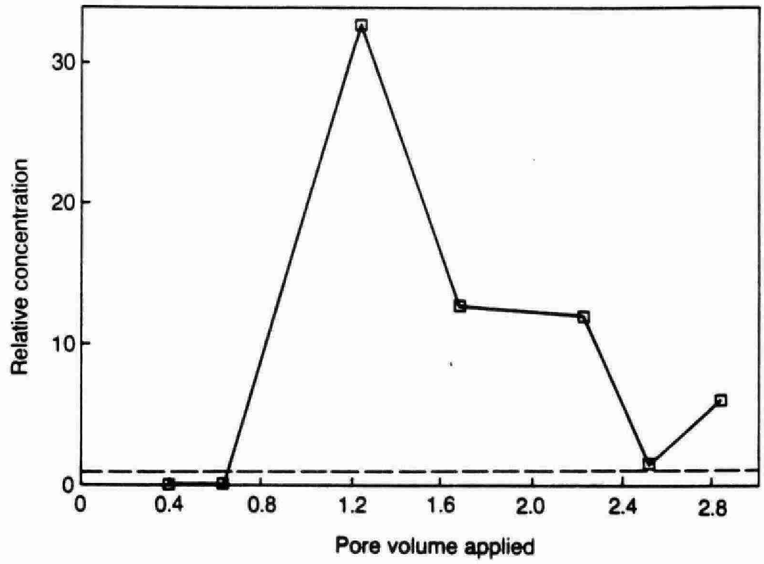


Figure 6.8. Elution pattern for NO_3^- given high untreated leachate application rate on previously sprayed soil.

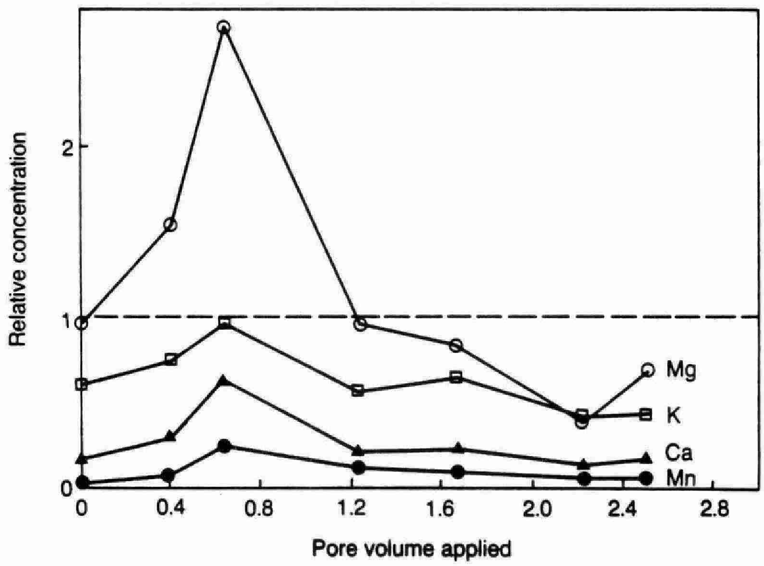


Figure 6.9. Attenuation (K,Ca,Mn) and elution-attenuation (Mg) patterns given high untreated leachate application rate on previously sprayed soil.

attenuated, with the basic cations (Ca, Mg, K) persisting much closer to the R.C. = 1.0 line even after 2.5 pore volumes. Peaks in the R.C. for most leachate constituents tended to occur after about 1.0 pore volume had been applied, but the precise maxima are somewhat obscured by the weekly periodicity of effluent sampling. These graphs also indicate that the attenuation-elution patterns of the Muskoka Lakes soils stabilize much more rapidly than with clay soils (Griffin et al., 1976).

Figure 6.10 shows that prolonged spraying has dramatically lowered the attenuation capacity of the Muskoka Lakes soil for K, as it has for most other cations. Only Fe attenuation appears to be essentially unaltered by past leachate application (Table 6.7). At the time of writing, no visible evidence of the formation of a Fe placic layer was apparent in any of the leaching columns containing previously unsprayed soil. Only destructive sampling will confirm the zone of Fe accretion if this ortstein does not manifest itself visibly.

Pretreatment of leachate with lime or activated carbon appears to have induced considerable elution of Mg and Mn relative to the untreated leachate columns, as has the rain water control. Thus, the application of these two relatively high pH pretreated leachates does not appear to have caused any major unforeseen effects on the soil or its attenuation capacity.

It was established in section 6.4.4 that the litter layer was critical to increasing the residence time of leachate above the groundwater table, thus enabling soil microbes to decompose the more noxious organic leachate constituents. Tables F.1 and F.2, however, would indicate that the forest litter layer and soil Ah horizon have limited influence on the attenuation of cations in solution since the effluent quality differs little between the intact and truncated columns at the intermediate application rate. It can only be speculated that the pH dependent C.E.C. in the upper soil zone of organic accumulation would become more important under higher application rates.

6.6 Conclusions

The spray irrigation rates recommended for the Muskoka Lakes site in earlier consulting reports have been set at 3.4 $mm \cdot d^{-1}$ for the peak summer months and 2.3 $mm \cdot d^{-1}$ in the spring and fall. These rates are apparently based only on mean daily potential evapotranspiration (Ep) rates during the spray season and a perceived requirement to balance the total annual leachate output with the spray area and spray season available. For example

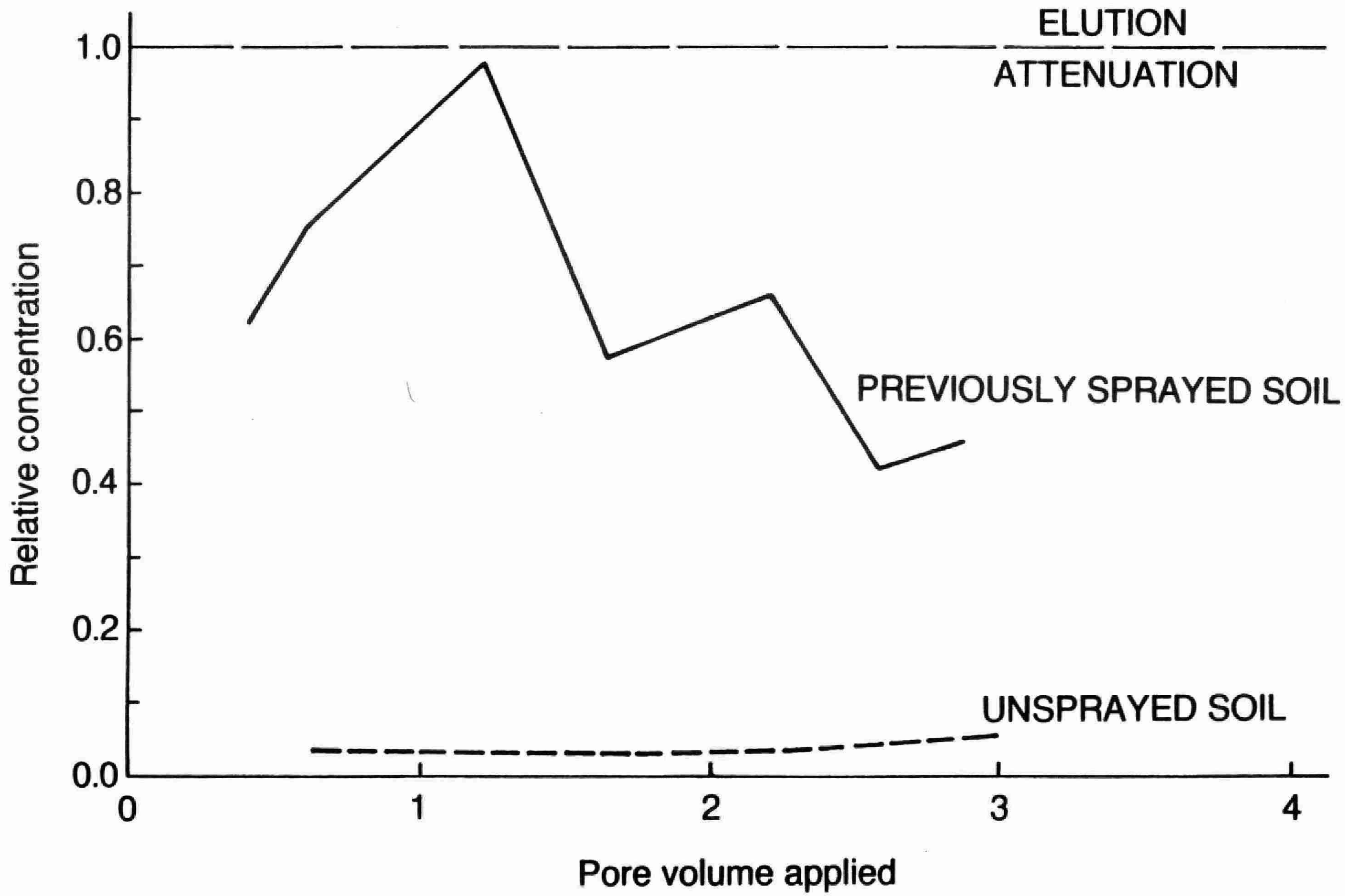


Figure 6.10. Attenuation patterns for K given high untreated leachate application rate on previously sprayed and unsprayed soil.

by using a modified energy budget approach to Ep estimation, the mean daily Ep rates for the months of May through October in 1986 were 3.31 mm, 3.29 mm, 3.58 mm, 3.34 mm, 1.66 mm and 1.20 mm, respectively. Both monthly and daily soil moisture budgets utilizing precipitation, Ep estimates and the soil storage capacity of the local soils have shown that the 1986 spray season was abnormally wet with few dry spells of any duration.

Superimposing an irrigation schedule under these types of conditions while attempting to avert serious soil waterlogging is very difficult. The reality is that a large volume of leachate must be handled during a limited spray season, and it is this more than any other factor which determines the spray application rates. A standard rain gauge placed 5 m from a spray nozzle position in 1986 frequently recorded daily application rates in excess of the recommended rates by many fold. With revised leachate generation rates of about $5.3 \times 10^4 \ell \cdot d^{-1}$ during the 1986 spray season, a mean daily irrigation rate of $1.23\ mm \cdot d^{-1}$ over the 4.3 ha spray area would have to be maintained to handle the daily accumulation, with no consideration given to rainfall amounts or the leachate generated during the winter season. Mean daily rainfall amounts of 3.32 mm, 4.24 mm, 3.35 mm, 3.10 mm, 5.82 mm and 3.34 mm from May to October in 1986 easily balance the mean daily Ep values cited above, leaving even an irrigation rate of 1.23 $mm \cdot d^{-1}$ clearly in excess of atmospheric demand. These excesses will be extreme in particular locations within the spray area due to the observation that greater than 80% of the leachate is applied to less than 20% of the 4.3 ha spray area with the current system (i.e. < 1 ha). Weather conditions in 1986 would certainly not have allowed for the effective implementation of a spray scheduling program more attuned to daily evapotranspirative conditions in the forest ecosystem. This will only be feasible during abnormally dry or more normal precipitation years.

Data gathered on the correspondence between seasonal precipitation amounts and leachate collected reaffirm the need to properly cap the landfill since most leachate originates from the unacceptably high rates of net infiltration through the existing surface liner. Leachate volume reduction is a key to the future success of spray operations at Muskoka Lakes. Other measures which will reduce both leachate volume and strength are i) continuous pumping from the culvert well or a drilled well in the same vicinity, ii) recirculation of the leachate through the landfill during the winter season, and iii) establishment of vegetation on the sand flats between the landfill and the collection trench.

The indigenous soils on site are coarse-textured, acidic Brunisols which evolve to form Podzols with higher pH and often a cemented podzolic B horizons after prolonged irrigation with landfill leachate. These soils possess a relatively low attenuation capacity for the basic cations (K, Ca, Mg) which are readily leached through the low C.E.C. subsoils. Iron, however, is very effectively removed from solution irrespective of the past spray history of the soil or the presence of the upper soil horizon of organic matter accumulation. Of concern are some of the more recalcitrant organics and NO_3^- which are likely to migrate rapidly to the local groundwater flow systems. Nitrates released from the mineralization and nitrification of organic detritus on the forest floor and leachate organics which are in excess of plant requirements will be rapidly leached under an irrigated environment.

7.0 Study of Biotic Stress from Leachate Irrigation

7.1 Preface

The visible damage observed in 1986 and previously in the tree and understory vegetation within the spray areas indicates that the practice of spray irrigating leachate into the hardwood forest ecosystem is effecting major stresses on the biota. The literature available on the subject is sparse and inconclusive as to the physiological cause of this stress. The most probable causes, however, are related to i) the phytotoxity of certain leachate constituents in direct contact with plant foliage and roots, and ii) the dramatic changes in the soil moisture and aeration regimes brought about by excessive and inequitable application rates of relatively high COD/BOD wastewater (i.e. 1600 and 892 ppm, respectively, in the settling lagoons).

Vegetative and soil microbial stresses were measured in the field using a number of techniques. Time-sequenced, low altitude infra-red photography of the spray area vegetation and other remotely-sensed information was used to broadly characterize the severity of these stresses at this point in time (section 7.2.1). Routine forest and ground cover mensuration techniques were also applied to ascertain the effect of leachate exposure on past growth patterns and survival (sections 7.2.2 and 7.2.3). More detailed microclimatic measurements assisted in isolating these effects at the level of individual leaves (section 7.2.4). The impact of leachate application on the soil microbial population was also investigated in some detail. Many of the leachate renovation processes and the availability of plant nutrients are intimately dependent on biochemical transformations in the most microbially-active soil zones (section 7.2.5). The lack of control over the spray program at Muskoka Lakes, both past and present, necessitated more controlled experimentation in a greenhouse environment to better monitor leachate contaminant uptake and plant physiological responses to leachate exposure (section 7.3).

Much of the above field activity was carried out within a series of permanent plots established early in the field season. Figure 6.2 shows the location of nine circular, 0.05 ha plots which were distributed in groups of three amongst the lightly sprayed area (spray area 3), the heavily sprayed area (spray area 1) and a control area on the north side of the hydro transmission corridor. The soil conditions and past spray history of each has been previously described in section 6.4.1. For all but the three control plots, the spray nozzle positions served as the plot centre point

(i.e. 12.6 m radius). The control plots were established around randomly selected centre points. The permanent plot locations were only finalized once it had been determined that variations in pre-spray stand structure, species composition and soil conditions were minimal.

7.2 Vegetative and Soil Microbial Stresses in Situ

7.2.1 Remotely-sensed Imagery and Canopy Measurements from Low Altitude

Thermodynamic principles dictate that the sum of the reflected, transmitted and absorbed light is equivalent to the incident light on a vegetative canopy (Wanjura and Hatfield, 1986). For vegetation under a stress sufficient to cause adverse physiological and metabolic responses, this energy balance will tend to shift toward a proportionately lower amount of reflected near infra-red (0.7-1.1 μm) radiation and a higher amount of reflected photosynthetically-active (0.4-0.7 μm) radiation.

7.2.1.1 Infra-red Reflectance of the Forest Canopy

Extensive use was made of time-sequenced, low altitude infra-red photography to broadly characterize the severity of tree vegetation stresses and the areal extent of noticeable forest decline within the 4.3 ha spray area. This technique is useful in identifying stress in vegetation before more visible symptoms become apparent, even at ground level. A series of four flights were made monthly between May and August, 1986 to allow qualitative assessment of the degree of vegetative stress, if any, as the spray season progressed. Figure 7.1 is representative of the imagery taken along the higher altitude flight lines while the lower altitude photographs allowed for stress assessment on individual trees.

This imagery demonstrated clearly that the forest decline is widespread in spray areas 1 and 2 (i.e. moderate and heavy spray histories) and that tree mortality is confined to small areas immediately surrounding the original six-nozzle spray system operative up to 1985. This is supportive of the data in section 6.4.3 which suggests that greater than 80% of the total leachate volume being sprayed each year is being applied to less than 20% of the total spray area. This is true of both current and past spray systems. Tree mortality was easily visible on the IR photography as gaps in the mauve-hued tree canopy. Forest decline was indicated by a shift in colour of individual tree crowns from mauve to pale pink or white (Murtha, 1972). This diminished IR reflectance is indicative of vegetation under

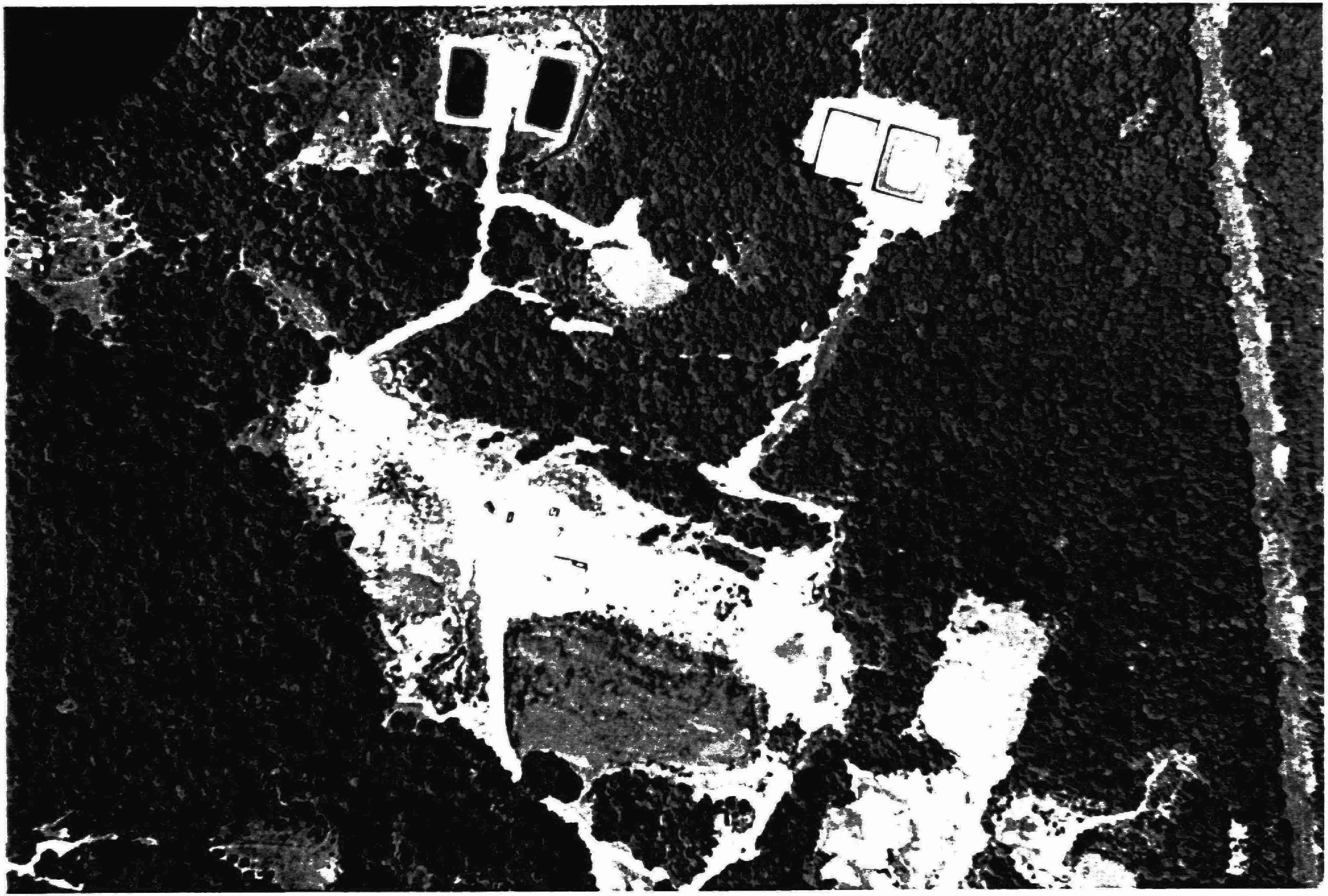


Figure 7.1. Low altitude, infra-red photograph of the Muskoka Lakes site taken in June, 1986.

considerable stress, although the precise cause cannot be interpreted from this imagery. The effect of leachate exposure on forest vegetation at the Muskoka Lakes site thus seems to be a threshold phenomenon, whereby a forest will go into general decline under appreciable leachate loading rates but will only expire once a critical application rate is exceeded. Given the prevailing spray distribution pattern noted in section 6.4.3, tree mortality would be expected to occur first in areas immediately adjacent to the spray nozzles where the irrigation rates are highest.

Very similar patterns were observed on aerial IR imagery taken in July, 1986 over two nearby land-based sewage wastewater treatment sites. The Huntsville Holiday Inn spray site has not been operational for some time even though the lagoon and remnants of the spray system itself still exist on site. Ground level observations confirmed the large gaps in the tree canopy captured by the IR photographs, sites now strewn with dead tree stems and a proliferation of the hydrophillic understory species Jewel-weed (Impatiens biflora L.). Some evidence of forest decline but no mortality was noted on the IR photographs taken at the still active Cleveland House sewage wastewater spray area. This aerial survey was not followed up by on-site inspection.

7.2.1.2 PAR Reflectance of the Forest Canopy

Measurements were made during the July 8, 1986 flight of photosynthetically-active radiation (PAR) reflectance from tree canopies within and outside of the boundaries of the Muskoka Lakes spray area. A quantum (PAR) radiometer was used from an altitude of 150 m above ground level. In a perfectly vertical upfacing position, the radiometer registered about 2000 $\mu E \cdot m^{-2} \cdot s^{-1}$ incoming PAR between 12:30 p.m. and 1:30 p.m. (i.e. clear skies, low humidity). PAR reflected from the forest canopy with the sensor in a vertical downfacing position was about 80 $\mu E \cdot m^{-2} \cdot s^{-1}$ and did not fluctuate significantly as the flight path crossed over the spray area from an unsprayed forest canopy. Thus, the anticipated relative increase in reflected PAR over the forested spray area could not be confirmed using a radiometer at the slowest airspeed and lowest legal altitude possible.

It is believed that foliar staining with leachate spraying in the understory will give rise to increased PAR reflectance at the expense of PAR absorptance (i.e. photosynthesis) and transmittance. No attempt was made to confirm this hypothesis in this study.

7.2.2 Understory Community Dynamics

The ground level observations reported above at the Huntsville spray site clearly showed significant encroachment of hydrophillic understory species into the now abandoned spray area. This prompted an ancillary study of understory vegetation community changes attributable to leachate spraying at the Muskoka Lakes site. Two of the more important factors that determine the regional and local distribution of vegetation species are the soil nutrient and the soil moisture-aeration regimes, both of which can be altered significantly by leachate irrigation.

A survey of ground cover, including small saplings, shrubs and herbaceous plants was conducted at the Muskoka Lakes site in mid-September. The purpose of the survey was to determine any variance in the composition of the ground cover, specifically the presence or absence of hydrophillic species amongst the heavily and lightly sprayed and control (non-spray) areas. The presence of hydrophillic species on rapidly drained, coarse-textured sites was thought to be a clear indication of a major shift in the soil moisture regime attributable to leachate irrigation.

The ground cover species within each of the nine previously established 0.05 ha permanent plots (Figure 6.2) were identified on site or specimens were gathered for later identification. These data were analyzed using the Mueller-Dombois and Ellenberg (1974) tabular classification system. This method is designed to group stands of similar species composition into vegetation units or types. A synopsis of the results is presented in Table 7.1 with the full species tabulation in Appendix G.

Table G.1 in Appendix G indicates that two vegetation units can be distinguished. These include areas which have been sprayed, irrespective of application rate, and those which have not been exposed to leachate irrigation. The "sprayed" vegetation unit can be identified by the presence of such species as shield fern, wild leeks, mountain-rice and sedges. The "unsprayed" areas will often have one or more of the spinulose wood-fern, willow, twisted stalk, hobble-bush and hepatica. This division is somewhat tenuous as, of the total of 32 species identified across all nine plots, 18 were located in more than one plot. Furthermore, of these 18 species, 10 were common to both sprayed and unsprayed plots. Twelve hydrophillic species were observed across all nine plots, of which 9 were identified within the heavily sprayed plots. Only 4 and 5 wet habitat species were

Table 7.1. Summary of the Mueller-Dombois and Ellenberg (1974) understory vegetation classification.

	Sampling Area								
	Heavy-spray plot no.			Light-spray plot no.			Control plot no.		
	1	2	3	1	2	3	1	2	3
Total number of understory species per plot	10	8	10	5	8	11	8	10	15
Total number of understory species by treatment (3 plots) relative to the total observed on all 9 permanent plots		17/32			15/32			19/32	
Total number of understory species preferring wet habitats per plot	5	3	3	1	2	3	2	3	1
Total number of understory species preferring wet habitats by treatment (3 plots) relative to the total observed on all 9 permanent plots		9/12			4/12			5/12	

found in the lightly sprayed and control areas, respectively (Tables B.1 and 7.1). Excessive spray irrigation has clearly induced soil moisture and nutrient conditions which are favourable to the encroachment and establishment of hydrophillic vegetation species.

7.2.3 Forest Growth and Productivity

7.2.3.1 Forest Mensuration Techniques

To gain a better understanding of both past and present effects of leachate irrigation on a mature forest ecosystem, canopy foliar samples taken from above the spray line and stem increment cores were collected within each of the nine 0.05 ha permanent plots established in the heavily sprayed, lightly sprayed and control areas (Figure 6.2).

Canopy foliage was sampled in triplicate for each species present within these plots three times through the growing season. These samples were then bulked into analytical samples representing the mid- (July and August) and late (October) growing season. The foliage was analyzed for levels of N, P, K, Ca, Mg, Mn, Fe, B and Zn. Analysis of variance was

performed on the data from two indicative species (i.e. beech and maple) to determine the effect of leachate application on nutrient uptake and accumulation by the mature trees.

Increment cores were taken from the stems of each maple and beech tree situated within the nine permanent plots and having a diameter greater than 10 cm. The computer-assisted T.R.I.M. (Tree Ring Increment Measurement) system was used to facilitate these measurements. An example of the T.R.I.M. output is presented in Appendix H. The data from individual trees of each species were bulked and an analysis of variance was performed to establish the degree of variation in tree vigor over the last 5 years which might be attributable to leachate exposure. Separate analysis on the annular rings from the last 12 and 7 years of growth revealed no significant difference in forest productivity amongst the nine plots prior to leachate application.

7.2.3.2 Forest Productivity Results

Changes in foliar nutrient concentration of the sampled trees reflect the influence of leachate application over the 1986 spray season (Table 7.2). Foliar concentrations of N and P were significantly higher in both maple and beech within the sprayed areas when compared to the unsprayed area (control). This is consistent with other published literature (Sopper and Kardos, 1979; Menser et al., 1979; Menser et al., 1983) and generally reflects the uptake of substantial quantities of these nutrient elements available in the spray-irrigated leachate. For the maple foliar samples, K and Mg levels increased significantly within the sprayed areas in comparison to those levels in the control. Concentrations of K and Mg did not vary between treatments in foliage sampled from beech trees.

Levels of Mn in the maple foliage, although markedly higher within the lightly sprayed area, declined in the heavily sprayed area when compared to the control. The availability of Mn to plants is greatly reduced as soil alkalinity increases (Foy et al., 1978). Table D.4 shows that pH increases substantially in the upper soil horizons with prolonged leachate applications (i.e. soil pit 5), thus reducing Mn in the soil solution in the heavily sprayed areas. Lighter applications over shorter periods of time on acidic, sandy soils, however, will cause increased Mn availability due to the reducing redox environment created by periodic waterlogging (Table 7.2). The mean Mn levels in beech foliage show a similar trend but the means are

Table 7.2. Canopy foliar analyses for maple and beech trees under different leachate irrigation treatments.

Tree Species	Spray Treatment	Elemental Foliar Analyses									
		N	P	K	Ca	Mg	Mn	Fe	B	Cu	Zn
		% $g \cdot g^{-1}$					ppm				
Maple	Heavy	2.61 a	0.11 a	1.13 a	1.65	0.26 a	760.0 c	1086.7	136.46	16.72	38.50 b
	Light	2.41 a	0.11 a	1.03 ab	1.67	0.27 a	1552.3 a	558.3	69.03	17.56	54.52ab
	Control	2.08 b	0.10 b	0.92 b	1.65	0.20 b	1093.6 b	316.8	57.03	18.12	63.66 a
Beech	Heavy	2.77 a	0.13 a	1.09	0.89	0.24	500.5	1540.2a	57.39a	21.18	40.64
	Light	2.67 a	0.13 a	1.12	1.20	0.25	802.7	421.6b	37.08b	88.89	77.37
	Control	2.30 b	0.10 b	1.10	1.30	0.23	547.9	357.7b	24.68b	20.50	37.36

Means with the same lower case letter are not significantly different at $P = 0.05$.

not statistically distinguishable. Zinc concentrations in maple leaves decreased appreciably within the heavily sprayed areas relative to levels observed in the control. Any trends observed in the mean maple foliar concentrations for the remaining elements (i.e. increasing Fe and B with higher leachate applications) were not statistically significant.

From analyses of the beech foliage, both Fe and B increased significantly within the heavily sprayed area relative to samples from the control. Foliage composition of K, Ca, Mg, Mn, Cu and Zn did not vary between treatments.

Table 7.2 provides substantial evidence of increased uptake and accumulation of many inorganic leachate constituents, particularly the macronutrients (N, P, K), within the biomass of the hardwood species investigated with leachate application. The forest vegetation is thus contributing substantially to the renovation of the wastewater by assimilating these substances before they migrate through the root zone in deep drainage water and reach the groundwater table. The actual extent to which the vegetation is attenuating the leachate is, however, difficult to estimate because the annual storage of nutrients in the woody tissue and the extent of recycling of nutrients in the litter layer of these forest soils are unknown (Sopper and Kardos, 1979).

Stem growth of both maple and beech over the past 5 years in the heavily sprayed area, as measured from increment borings, was not significantly affected by the application of leachate (Table 7.3). One year of spraying similarly had no effect in the lightly sprayed area. This is not an unreasonable finding. Under normal conditions, plant assimilate resources are allocated to diameter growth only after the demands by new foliage and buds, new roots and canopy/stem storage have been met. Thus, environmental changes causing stress in trees should be first detected in foliage, fruit structures and buds (Table 7.2). Extended leachate application may eventually decrease annual ring width but this tenet must be qualified by the possible additive growth effect due to the nutrient content of the leachate.

Table 7.3. Effect of leachate application on beech and maple stem growth.

Vegetation Species	Spray Treatment	*Mean Growth on Year Before Present (mm)		
		Year 1	Year 5	Year 7
	Control	1.635	1.633	1.622
Maple (Mh)	Heavy	1.583	1.598	1.578
	Light	1.823	1.868	1.828
	Control	1.866	1.948	1.856
Beech (Be)	Heavy	1.255	1.323	1.773
	Light	1.813	1.716	1.327

* Means not significantly different at P = 0.01.

7.2.4 Forest Transpiration

7.2.4.1 Effects of Leachate Irrigation on Plant Transpiration

A critical aspect of this study was to obtain some direct measure of the physiological response of the predominant vegetation species on site to spray irrigation of leachate. This objective was achieved via direct porometric measurements of the transpiration rate and diffusive (stomatal) resistance of plant leaves both in the understory and forest canopy.

The LI-1600 Instrument

All measurements were taken with a LI-COR LI-1600 Steady State Porometer. The LI-1600 is a portable null balance porometer used to measure water loss and diffusive resistance from transpirating surfaces (LI-COR Inc., 1984). In addition the porometer measures leaf temperature, relative humidity, and incoming photosynthetically-active radiation (PAR).

The porometer consists of two parts; the "readout-control console" and the "sensor head" which is clamped onto the leaf and holds the sampling cuvette. The basic operating principle is that the rate at which dry air (0% relative humidity) is added to the cuvette to balance the increase in humidity due to leaf transpiration and to keep the relative humidity at the balance point can be used to determine stomatal resistance (Beadle et al., 1985). The rate of air flow is semi-automatic, controlled by a null adjustment valve (manual) and an internal air flow controller. Diffusive resistance and transpiration rate are instantaneously calculated under ambient environmental conditions from the measurement of leaf temperature, cuvette temperature, relative humidity and dry air flow rate.

Leaf stomatal aperture is a function of several environmental factors including light levels, wind speed, CO_2 concentrations, relative humidity, ambient air temperature, leaf temperature and leaf water potential. The LI-1600 is designed to minimally disturb these factors within practical limits, although the design does not attempt to prevent the alteration of the wind environment. The boundary layer resistance is small as long as the cuvette is properly aspirated (Beadle et al., 1985). Incident PAR is not diminished by the instrument.

Despite the sophistication of instrument design, the diffusive resistance computed is not a precise measure of the actual diffusive resistance of the leaf in a natural state. This is due in part to the length of time the leaf must be clamped onto the sensor head before an equilibrium is reached. The longer the sampling time, the more the measured diffusive resistance is lowered by alteration of the leaf environment. In addition, the leaf sample may not form an airtight seal due to holes or veins in the leaf. Therefore, the air in the cuvette may not be at ambient humidity and the exchange of water vapour with the environment will affect the diffusive resistance reading. These limitations have been noted by Gee and Federer (1972) who view the porometer as a semi-quantitative instrument. Design modifications in the recently developed LI-6200 portable photosynthesis instrument allow these same measurements to be made with virtually no change in the environment of the plant tissue.

Pre-field Calibration and Field Measurements

As part of the initial operating preparation of the LI-1600, both dessicant packs of silica gel were dried thoroughly before using the instrument. This was done by heating the silica gel in an oven at 175°C for one hour. The instrument was then pre-calibrated to 2% and 94% relative humidity to bring the instrument within the proper calibration range. A saline solution procedure was then used to calibrate the relative humidity function of the porometer. $MgCl_2$ and NaCl were used to prepare a saturated salt solution and the sensor head was placed over the two solutions at constant temperature. These salts were used to correspond to the operating range of the relative humidity function. $MgCl_2$ provides a low humidity reference point of 32.7% and the NaCl a high humidity calibration of 75.1%.

The first step in the measurement process was to acclimate the instrument to ambient temperature and humidity conditions near the leaf sample. The leaf was then clamped into place on the sensor head. Once a

steady state or null balance was achieved, one set of parameters was recorded. A null balance is achieved when the humidity in the cuvette is maintained at the ambient air humidity value. The time taken to attain a null balance averaged 10 to 20 seconds.

Data were collected at the Muskoka Lakes site from sugar maple leaves at two strata in the forest cover profile and from beech leaves in the understory only. The understory was sampled at heights of 0.5 - 1.0 m and the mid-canopy at heights of 10-15 m. A tree pruner was used to sample leaves in the forest canopy. Branches 1 to 3 cm in diameter were cut from the canopy and porometer measurements were made immediately at ground level. This method of sampling was considered appropriate due to the impracticality of setting up scaffolding at several sampling locations and the belief that none of the measured parameters would be significantly affected given the rapidity of measurement.

Measurements were taken within the nine permanent plots (Figure 6.2) which represent different leachate application treatments and an unsprayed control. Supplementary readings were also taken in the moderate spray area (i.e. spray area 2). All measurements were made during the period of August 12 to 31, 1986. The time of data collection was confined to within four hours of solar noon. The winds were light to moderate and skies were predominantly clear on all days that measurements were made.

Results

The porometer measurements obtained in the field were referenced against similar measurements for the same species as found in the literature. Turner (1969) and Jurik (1986), using independent methods, both found that leaves in the understory have higher diffusive resistances than leaves at the top of the forest canopy, due apparently to differences in the radiation regimes. This observation was confirmed from measurements taken at the Muskoka Lakes site. Gee and Federer (1972), in a porometer-based study of hardwood forests, found diffusive resistance values for beech in the range of 3-5 $s \cdot cm^{-1}$ and 2-3 $s \cdot cm^{-1}$ for striped and sugar maple seedlings. These are consistent with the values in Table 7.4 for the unsprayed vegetation, but these resistances increase rapidly with leachate irrigation both in the understory and mid-canopy.

Porometer measurements taken on the same day under relatively constant atmospheric conditions for the beech understory (Table 7.4) show a significant increase in diffusive resistance and decline in transpiration

Table 7.4. Measurements of diffusive resistance and transpiration rates of beech and maple vegetation exposed to leachate spraying.

Vegetation Species (sampling date)	Spray Treatment	Sample Size (n)	Means (Standard Errors in Brackets)				
			Diffusive Resistance	Transpiration Rate	Photosynthetically-Active Radiation	Relative Humidity	Leaf Temperature
			$s \cdot cm^{-1}$	$\mu g H_2O \cdot cm^{-2} \cdot s^{-1}$	$\mu E \cdot m^{-2} \cdot s^{-1}$	%	°C
Beech Understory							
(Aug.31/86)	Control (non-spray)	29	2.78 b (0.075)	4.127 a (0.091)	13.5 (1.25)	34.69 (0.037)	21.0 (0.05)
(Aug.31/86)	Moderate Spray	34	3.47 a (0.110)	3.422 b (0.107)	176.9 (55.47)	34.77 (0.047)	21.3 (0.11)
(Aug.31/86)	Heavy Spray	29	3.69 a (0.140)	3.207 b (0.117)	21.2 (4.56)	34.54 (0.019)	21.0 (0.02)
Maple Understory							
(Aug.13/86)	Control (non-spray)	95	3.21 c (0.094)	3.776 a (0.106)	29.9 (4.99)	38.47 (0.50)	22.0 (0.06)
(Aug.12/86)	Light Spray	92	5.62 b (0.296)	1.765 b (0.079)	47.9 (13.50)	49.49 (0.31)	19.0 (0.08)
(Aug.18/86)	Moderate Spray	117	5.20 b (0.209)	1.557 c (0.067)	54.9 (15.74)	63.79 (1.02)	22.6 (0.16)
(Aug.12/86)	Heavy Spray	83	6.94 a (0.290)	1.564 c (0.069)	219.6 (37.92)	42.01 (0.01)	19.3 (0.05)
Maple Mid-canopy							
(Aug.31/86)	Control (non-spray)	50	2.43 b (0.046)	4.665 a (0.079)	11.4 (1.81)	34.54 (0.02)	20.8 (0.02)
(Aug.31/86)	Moderate Spray	78	2.58 b (0.048)	4.253 b (0.896)	67.9 (17.85)	34.80 (0.02)	19.8 (0.25)
(Aug.18/86	Heavy Spray	125	4.76 a (0.186)	2.677 c (1.107)	135.1 (22.31)	49.17 (0.61)	22.1 (0.34)

Means with the same lower case letter are not significantly different at P = 0.01.

rate with leachate irrigation. No significant difference was indicated, however, between the different spray regimes. A similar trend is evident for maple seedlings even under variable ambient environmental conditions on different sampling dates. In this instance, increasing irrigation amounts amongst the three spray areas have induced significantly more pronounced adverse physiological responses in the maple understory. Even the canopy foliage of the maple species shows a marked depression in transpiration with progressively greater leachate loadings.

Intuitively, irrigated vegetation should have higher transpiration rates due to lessened soil resistances (i.e. higher unsaturated hydraulic conductivity) to root uptake of water. The leaf porometer measurements in Table 7.4 demonstrate, in fact, that the opposite is true at the Muskoka Lakes site. The most probable explanation for this important observation is the known negative effect that soil waterlogging can have on root function, including both water and nutrient uptake. Even though ample or excessive water exists in the soil, the root system cannot extract it efficiently. Insufficient oxygen in the low air-filled porosity of an over-irrigated soil inhibits proper root respiration and leads to root disfunction and ultimately death. In addition to excess irrigation, the high COD/BOD content can also contribute to a poor soil aeration condition as available oxygen in the soil air is utilized by microbes to breakdown these organic contaminants. The rapid increase in diffusive resistance with leachate irrigation will almost certainly bring with it much reduced photosynthesis rates in the vegetation due to increased resistance to CO_2 uptake and fixation in the leaves.

7.2.5 Soil Microbial Community Dynamics

7.2.5.1 Preface

A study was undertaken to assess the effects of leachate application by spraying upon two indices of forest soil health: soil microbial biomass and respiration. Soil respiration, or CO_2 evolution, has been shown to be relatively sensitive to different forms of toxic contamination (Salonius and Mahendrappa, 1979; Skujins et al., 1986). Similarly, soil microbial biomass is a good index of soil pollution (Nordgren et al., 1986). Both are readily measurable in the field and can provide information on decomposition rates and forest nutrient cycling processes as well as the effect of leachate application on these transformations.

7.2.5.2 Experimental Procedures for Soil Microbial Biomass and Respiration Measurements

Soil respiration was determined within the control and irrigated (light, heavy) permanent plots (Figure 6.2) in situ using the alkali absorption technique described and evaluated by Edwards (1982), and recently used successfully by Schlenter and Van Cleve (1985) and Gordon et al. (in press). Carbon dioxide evolved over a 24 hour period was captured by placing open tins of soda lime under inverted closed plastic containers imbedded into the forest floor. The collection system met all requirements established by Edwards (1982). The CO_2 evolved from the forest soil was calculated from the weight gain of the soda lime ascertained after oven-drying the absorbent for 24 hours at 105°C and was expressed as g CO_2 evolved$\cdot m^{-2}hr^{-1}$. The efflux of CO_2 was measured weekly from July 22 to September 25, 1986 at five random locations (replicates) within the permanent plots representing unsprayed, lightly sprayed and heavily sprayed conditions. An additional measurement to ascertain low-temperature effects was made on October 21, 1986. One "control" capped container of soda lime was employed during each weekly determination to measure any minute gains in CO_2 which might occur during transportation of the containers from the field to the laboratory. Weight measurements were multiplied by 1.41 to account for water which is released from the soda lime when CO_2 is absorbed (Edwards, 1982).

Soil microbial biomass was determined by a modification of the method developed by Jenkinson and Powlson (1976) (Dr. P. Voroney, pers. comm.). Soil samples were collected at three depths (i.e. 0-5 cm, 5-10 cm, 10-15 cm) from each of the heavily sprayed, lightly sprayed and control areas. Ten samples from each depth were aggregated to provide the analytical sample. The soils were incubated aerobically for one week at room temperature in loosely sealed plastic bags in order to allow the stress of sampling on soil activity to subside. Subsequently, the samples were either analyzed immediately or stored at 4°C. Earthworms, stones and large pieces of plant material were removed by hand and the soil was dispensed into 25 g samples for biomass determination.

The biocidal fumigation procedure used was as described and evaluated by Jenkinson and Powlson (1976). Erlenmeyer flasks containing 25 g of moist

---

Dr. R.P. Voroney, Assistant Professor of Soil Science, Department of Land Resource Science, University of Guelph, Guelph, Ontario.

soil were placed in a desiccator containing wet filter paper and 50 ml of alcohol-free, hydrocarbon-stabilized liquid chloroform. The desiccator was evacuated and left for a 24 hour period at 25°C until the fumigation was complete. Non-fumigated samples were similarly incubated for 24 hours at room temperature and served as the control. After fumigation, the soil solution was extracted with 25 ml of K_2SO_4 for 1 hour, filtered (Whatman no. 41) and the extracts stored at 5°C until analysis the following day. A white precipitate, presumably $CaSO_4$ (Brookes et al., 1985), formed upon storage of the soil extracts and was removed prior to analysis by additional filtrations. Total organic carbon (TOC) was measured on an auto-analyzer. The microbial biomass present was estimated as the difference in TOC between the fumigated and non-fumigated (control) samples.

Both soil respiration and microbial biomass data were evaluated by a one-way analysis of variance and multiple range tests (e.g. Duncan's). Due to the high range in the microbial biomass data, the raw data were transformed to natural logarithms prior to analysis.

7.2.5.3 Soil Respiration Results

Seasonal patterns of soil respiration for the lightly sprayed, heavily sprayed and control areas are presented in Figure 7.2. Estimates ranged from 0.23 g CO_2 evolved $\cdot m^{-2} \cdot hr^{-1}$ in mid-July to late season values of about 0.1 g CO_2 $\cdot m^{-2} \cdot hr^{-1}$ in mid-September and October. The values of soil respiration obtained from the control area (Table 7.5) are comparable to those described by Anderson (1973), who reported CO_2 evolution from two deciduous woodland soils, and Weber (1985), who reported soil respiration values for jack pine sites in eastern Ontario. The respiration estimates derived from the control are also within the range of respiration rates from other temperate mixed hardwood forests (Singh and Gupta, 1977).

Table 7.5. Effect of landfill leachate spraying on soil respiration at the Muskoka Lakes landfill site.

Spray Treatment	Soil Respiration	
	g $CO_2 \cdot m^{-2} \cdot hr^{-1}$	
Control	0.1249	a
Heavy	0.1390	ab
Light	0.1546	b

Means with the same lower case letter are not significantly different at P = 0.05

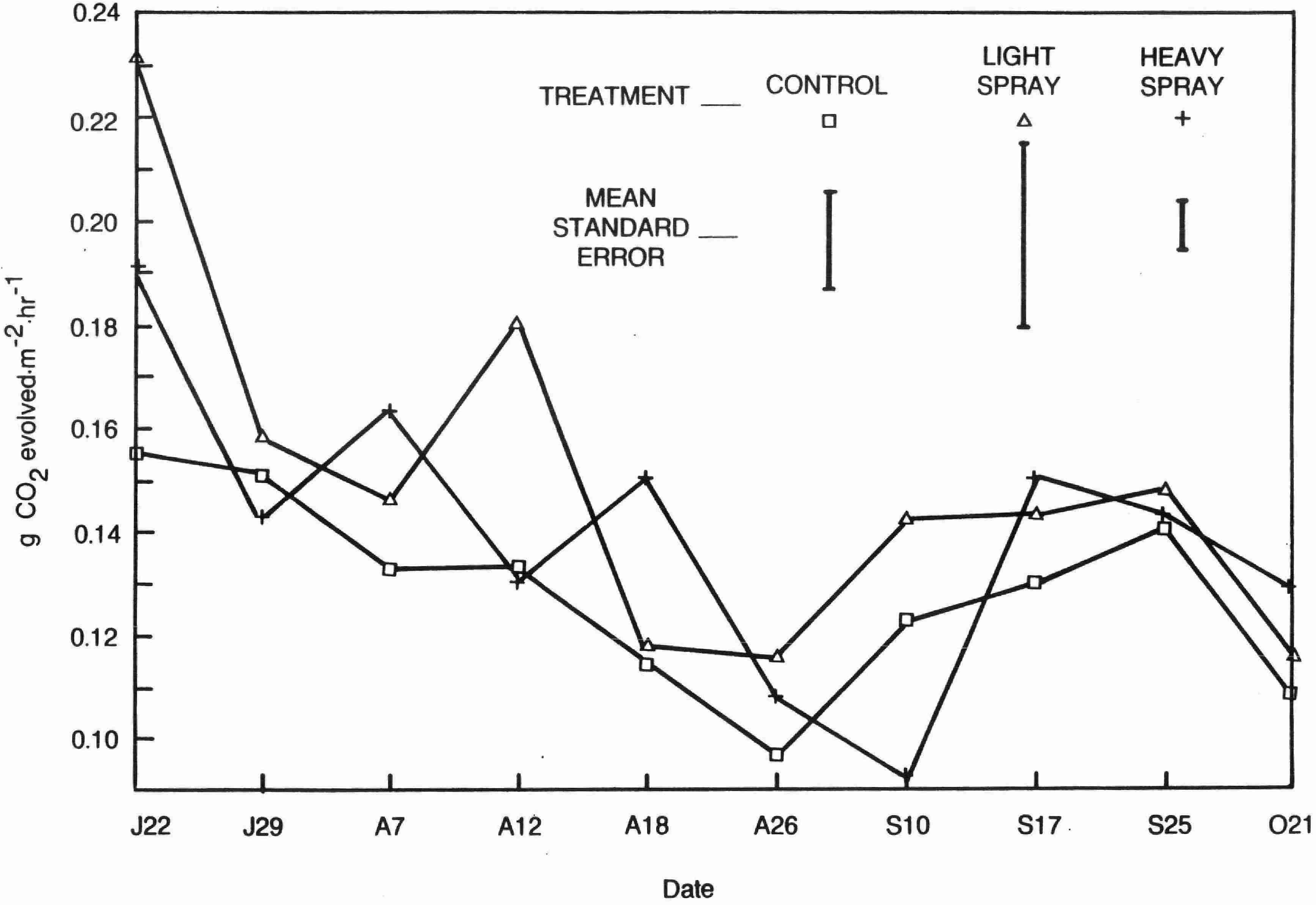


Figure 7.2. Seasonal soil respiration in areas sprayed with landfill leachate vs. an unsprayed control.

A one-way analysis of variance and multiple range testing indicated that the lightly sprayed area had significantly higher respiration rates than the control (Table 7.5). The decomposition of organic matter is the primary source of CO_2 evolution from the soil, except for the approximate 30% contribution from live root metabolism (Singh and Gupta, 1977). It may then be hypothesized that the increased CO_2 evolution,an index of microbial activity, within the lightly sprayed area is due primarily to the application of the leachate. Respiration in the heavily sprayed area did not differ from the lightly sprayed or control areas (Table 7.5). It is likely that more frequent application of leachate at higher rates and over a longer period of time (i.e. 1980-86) would cause chronic waterlogging with a concomitant decline in organic matter decomposition rates under these anaerobic conditions (Miller, 1979). Elemental toxicities to microbes may also be a factor due to the high levels of Fe in the sprayed leachate and the increased availability of soil Mn under low pH and waterlogged conditions.

7.2.5.4 Soil Microbial Biomass Results

Microbial biomass - TOC patterns from each treatment area and soil depth for the August and October sampling dates are illustrated in Figure 7.3 and the data are presented in Table 7.6. As expected, the soil microbial biomass decreased between summer and fall and with depth in the soil (Miller, 1979; Clarholm and Rosswall, 1980; Ferderle et al., 1986). The relationship that was unknown and formed the basis for the experiment was the effect of leachate exposure on microbial biomass. In general, the biomass-TOC measures from the control (Table 7.6) are within an acceptable range of comparably obtained values from other temperate woodlands (Brookes et al., 1984; Jenkinson and Powlson, 1976).

As the season progressed and soil temperatures dropped, a marked decrease in measurable biomass-TOC occurred (Table 7.7). Both the heavily sprayed and lightly sprayed areas exhibited this trend, but values obtained in the control area were not significantly different on the two sampling dates. A one-way analysis of variance and subsequent range testing also revealed that the microbial biomass-TOC measured on August 22 was significantly lower in the heavily sprayed area than in the control or lightly sprayed area. No difference existed between the lightly sprayed

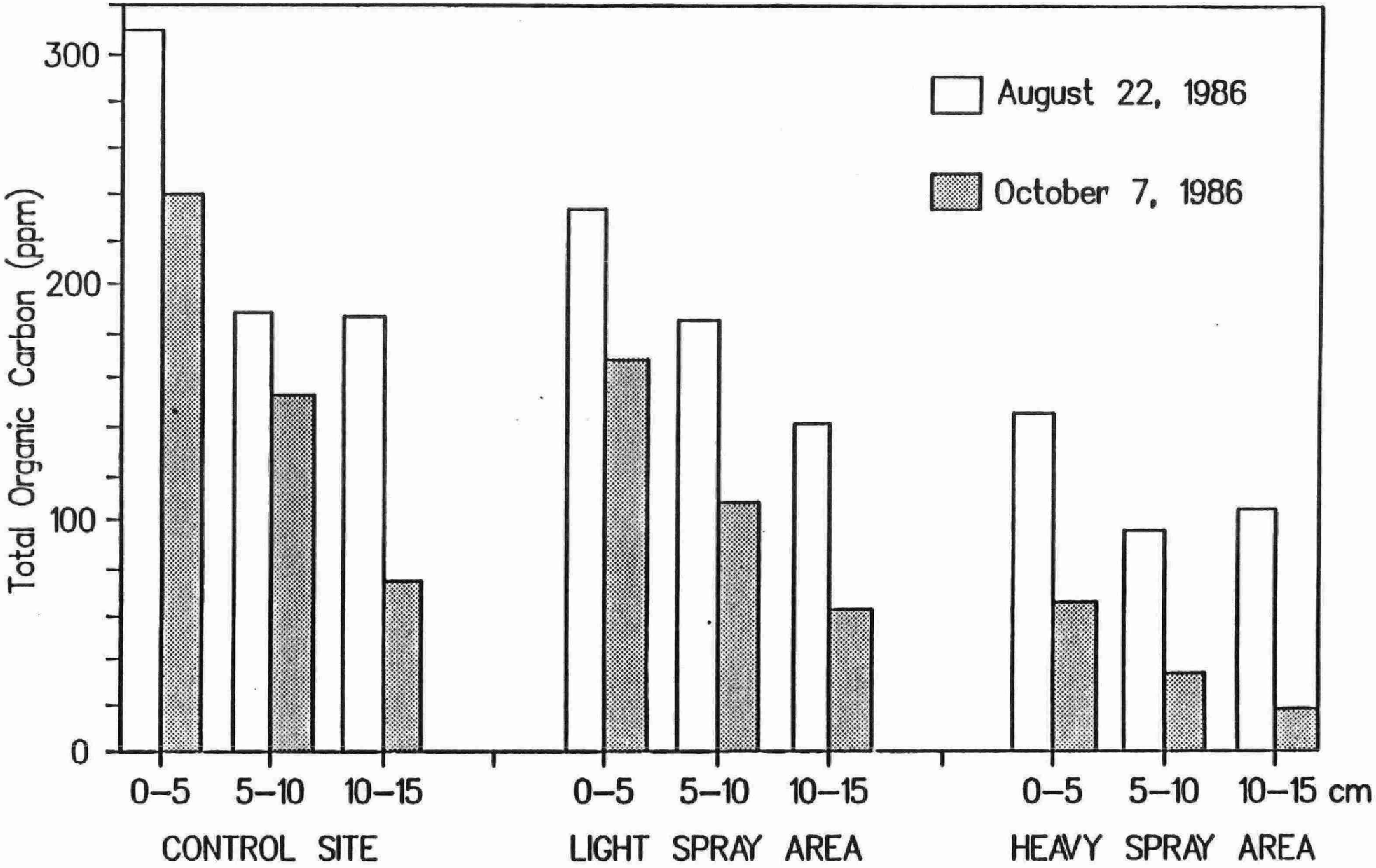


Figure 7.3. Total organic carbon from soil microbial biomass after biocidal fumigation with chloroform.

Table 7.6. Total organic carbon from soil microbial biomass after biocidal treatment with chloroform.

Spray Treatment	Date	Soil Depth	Microbial Biomass - TOC Mean	Standard Error	Range	ln-transformed data Mean	Standard Error
		cm	ppm	ppm	ppm		
Control	Aug.22	0-5	309.86	24.56	277-370	5.73	0.09
	Aug.22	5-10	195.06	52.75	70-285	5.11	0.44
	Aug.22	10-15	186.77	20.32	160-236	5.21	0.13
	Oct.07	0-5	239.44	23.96	200-297	5.46	0.12
	Oct.07	5-10	153.29	12.37	151-180	5.02	0.10
	Oct.07	10-15	74.13	5.63	64-87	4.30	0.09
Heavy	Aug.22	0-5	144.60	19.97	97-177	4.94	0.18
	Aug.22	5-10	96.61	13.99	63-118	4.53	0.20
	Aug.22	10-15	104.30	11.90	89-133	4.62	0.13
	Oct.07	0-5	65.19	10.47	42-86	4.13	0.21
	Oct.07	5-10	34.34	0.33	34-35	3.54	0.01
	Oct.07	10-15	19.79	3.96	13-29	2.92	0.24
Light	Aug.22	0-5	232.38	22.70	203-288	5.43	0.11
	Aug.22	5-10	186.10	11.74	158-204	5.22	0.08
	Aug.22	10-15	141.44	13.15	114-170	4.94	0.12
	Oct.07	0-5	168.03	21.26	119-207	5.10	0.17
	Oct.07	5-10	108.66	7.31	90-119	4.68	0.09
	Oct.07	10-15	61.71	12.43	34-86	4.05	0.28

Table 7.7. Effect of landfill leachate spraying on soil microbial biomass-TOC averaged over three soil depths on two sampling dates.

Spray Treatment	Date (1986) August 22	October 7
	---------------- ln ppm	--------------------
Control	5.35 a	4.93 a
Light	5.20 a	4.61 b
Heavy	4.70 b	3.53 c

Means with the same lower case letter are not significantly different at P = 0.05 (data is ln-transformed)

area and the control (Table 7.7). On October 7, all treatments were significantly different with the biomass-TOC increasing in order of heavily sprayed, lightly sprayed and control.

The effect of leachate irrigation on microbial biomass-TOC at 3 depths (0-5, 5-10 and 10-15 cm) for both sampling dates was similarly evaluated by ANOVA testing (Table 7.8). Microbial biomass-TOC exhibited similar trends with depth, although slight differences occurred between treatments. In the control area, biomass-TOC measured at the 0-5 cm depth was significantly greater than the measurement obtained at 10-15 cm. The values obtained from the 5-10 cm depth, however, did not differ significantly from those immediately above or below within the soil profile. Biomass-TOC within the lightly sprayed area was similar for the 0-5 cm and 5-10 cm depths but decreased significantly at 10-15 cm. Within the heavily sprayed area, the biomass-TOC measured was similar across all three depths. This could be a function of the higher levels of particulate and soluble organic material loaded into spray area 1 over the last 6 years. Table D.2 shows much increased organic matter content in the pit 5 soil profile indicating increased availability of substrate with depth relative to unsprayed areas.

Table 7.8. Effect of landfill leachate spraying on soil microbial biomass-TOC at three soil depths (August - October, 1986).

Spray Treatment	Soil Depth (cm)		
	0-5	5-10	10-15
Control	5.60 a	5.07 ab	4.76 b
Light	5.27 a	4.95 a	4.50 b
Heavy	4.54 a	4.14 a	3.78 a

Means with the same lower case letter are not significantly different at P = 0.05 (data is ln-transformed)

7.3 Vegetative and Soil Microbial Stresses in an Atmospherically-Controlled Environment

7.3.1 Maple Seedling Response to Leachate Applications

7.3.1.1 Experimental Procedures for Greenhouse Seedling Trials

Exploratory greenhouse trials were designed to investigate the nature and severity of any stresses imposed on maple seedlings by the application of untreated and pretreated leachate. Red maple (<u>Acer rubrum</u>) seedlings

were obtained from the Orono Provincial Forestry Station and were planted in 10 cm diameter pots containing a mixture of commercial potting soil and soil collected from an unsprayed area at the Muskoka Lakes site. This species was utilized in lieu of sugar maple, which is dominant on site, because of its higher tolerance to wetter soil moisture regimes. The seedlings were allowed to acclimatize to greenhouse conditions for about two months before leachate application commenced. Three treatment replicates of 5 plants each were established in the greenhouse in a modified randomized complete block design (i.e. 195 seedlings in total).

A volume of approximately 80 ℓ of leachate was collected weekly from the collection trench on site and was transported in closed containers to the greenhouse. The leachate was then divided into three equal volumes whereupon one lot was lime-treated (2.0 g $Ca(OH)_2 \cdot \ell^{-1}$), one lot was passed through an activated carbon column (Calgon Filtrasorb 300) and one lot was left untreated. Both treated and untreated leachate and a deionized water control treatment were applied to the pots for a seven week period at the rate of approximately 175 $m\ell \cdot wk^{-1}$ (i.e. 3.0 $mm \cdot d^{-1}$). The irrigation treatments descriptions are given in Table 7.9.

Table 7.9. Summary of leachate treatment applications in the greenhouse seedling trials.

Treatment No.	Mode of Application	Dilution Factor	Liquid Applied
1	soil and plant	-	deionized water
2	soil and plant	full strength	untreated leachate
3	soil and plant	diluted 50% with deionized water	untreated leachate
4	soil and plant	diluted 75% with deionized water	untreated leachate
5	soil and plant	full strength	lime-treated leachate
6	soil and plant	diluted 50% with deionized water	lime-treated leachate
7	soil and plant	diluted 75% with deionized water	lime-treated leachate
8	soil and plant	full strength	carbon-treated leachate
9	soil and plant	diluted 50% with deionized water	carbon-treated leachate
10	soil and plant	diluted 75% with deionized water	carbon-treated leachate
11	soil only	full strength	untreated leachate
12	soil only	diluted 50% with deionized water	untreated leachate
13	soil only	diluted 75% with deionized water	untreated leachate

Plant height and stem diameter were measured prior to the initiation of the leachate irrigation treatments. After the seven week irrigation period, plant height and diameter were remeasured as well as the levels of N, P, K, B, Ca, Cu, Mg, Mn, Zn and Fe present in the leaves. The root systems and stems were also harvested and bagged separately for possible future analysis and investigation of partitioning of leachate constituents in plant tissue.

The above trials involved irrigation of all pots at rates consistent with the evapotranspirational demand in a greenhouse environment. An ancillary experiment was carried out to observe the response of sugar maple seedlings to soil waterlogging with leachate. Sugar maple seeds obtained from the Provincial Seed Station at Angus were stratified, germinated and transplanted into 7.5 cm diameter pots. After an acclimatization period of about 1 month, initial seedling heights were measured. The seedling root systems were then subjected to several cycles of waterlogging conditions with untreated leachate or deionized water for a duration of 2 or 24 hours. After five such treatment cycles, each followed by a drainage period, seedling height was measured and the leaves were destructively sampled and analyzed for the same ten nutrient elements as cited above in the larger seedling experiment.

7.3.1.2 Foliar Analysis Results

Plant growth was not significantly influenced by leachate exposure at the application rate used (i.e. 3 $mm \cdot d^{-1}$). Specifically, seedling height over a 7 week period was not significantly affected by application of leachate relative to the water control. Stem diameter, however, was positively affected by applications of both untreated and lime-treated leachate diluted 75% by deionized water. This increase in diameter growth relative to the other treatments is difficult to explain and may be purely coincidental. At the time of seedling harvest in October 1986, there were few visible symptoms of plant stress from leachate exposure in any of the replicated treatments.

Table 7.10 reports concentrations of major and trace nutrient elements in the foliage of red maple seedlings treated with either water (control) or varying concentrations of untreated, lime-treated and carbon-treated leachate as listed in Table 7.9. Comparison of nutrient concentrations with those found in the literature is difficult due to several inherent sources

Table 7.10. Concentrations of the major and trace nutrient elements in red maple seedlings irrigated with treated and untreated leachate.

Treatment No.	Foliar Elemental Analysis									
	N	P	K	Ca	Mg	Mn	Fe	B	Zn	Cu
	% $g \cdot g^{-1}$					ppm				
1	1.19 bcd	0.36 ab	0.65	1.880 ab	0.48	2268.9	764.2 bc	83.55 ab	201.6	31.14 ab
2	1.33 ab	0.32 abc	0.77	1.800 ab	0.49	1659.7	2235.4 a	94.02 a	161.0	24.67 abc
3	1.18 bcd	0.17 c	0.74	1.689 abc	0.41	1796.9	1902.5 a	66.11 bc	158.5	20.62 bc
4	1.09 cd	0.23 bc	0.78	1.591 bcde	0.37	1966.3	1073.2 b	64.83 bc	166.8	20.85 abc
5	1.32 abc	0.32 abc	0.85	1.939 a	0.46	1862.2	626.7 c	81.21 ab	178.9	18.43 c
6	1.30 abcd	0.27 bc	0.74	1.658 abcd	0.43	1977.7	470.7 c	63.23 bc	173.2	18.67 c
7	1.07 d	0.38 ab	0.73	1.653 abcd	0.43	1875.0	500.9 c	65.18 bc	190.1	21.33 abc
8	1.29 abcd	0.33 abc	0.73	1.645 abcd	0.50	1896.8	681.0 c	60.54 bc	165.3	24.29 abc
9	1.15 bcd	0.25 bc	0.81	1.208 f	0.37	1695.7	656.5 c	50.42 c	162.5	21.17 abc
10	1.22 bcd	0.48 a	0.77	1.386 cdef	0.45	1976.1	581.7 c	63.20 bc	170.9	32.27 a
11	1.48 a	0.28 bc	0.70	1.345 edf	0.46	1901.7	633.3 c	74.22 ab	163.9	26.00 abc
12	1.33 ab	0.30 bc	0.65	1.319 ef	0.46	2147.5	516.9 c	77.71 ab	196.1	26.67 abc
13	1.15 bcd	0.21 bc	0.63	1.492 cdef	0.40	2091.8	519.1 c	77.07 abc	174.1	26.67 abc

Means with the same lower case letter are not significantly different at P = 0.05.

of variation. These include temporal variations in foliar nutrient status (i.e. annual, seasonal, diurnal) as well as the effect of tree age, foliage age, foliage exposure, nutrient availability and soil property variations (Turner et al., 1978). Despite these many sources of variation, N, P and K concentrations observed in the water irrigated plants are all within the sufficiency range (Sopper and Kardos, 1979; Lozano and Morrison, 1981; Ricklefs and Matthew, 1982; Menser et al., 1983).

Iron and nitrogen were among the nutrient elements which showed noticeable differences in foliar concentrations between leachate and water irrigated seedlings (Table 7.10). Both full and half strength untreated leachate increased foliar Fe levels considerably in treatments 2 and 3 relative to treatment 1 (Table 7.9) since the leachate was applied to both the soil and the plant foliage. This simulation of spray irrigation clearly resulted in appreciable Fe precipitation on the leaf surfaces but no significant Fe uptake since treatments 11 and 12 (i.e. untreated leachate applied to soil only) showed no difference in Fe levels compared to treatment 1. Since much of the Fe was removed through pretreatment with lime and activated carbon (Table 6.8), treatments 5-10 are also no different from treatment 1 in Fe levels. In comparison to the water-treated seedlings, foliar N increased significantly in plants which received applications of full strength, untreated leachate to the soil only. Substantial quantities of N are thus made available to plants through leachate application supporting the view that MSW leachates have good potential for recycling in forest ecosystems. Although not statistically defensible, the mean N levels tended to decline as untreated or pretreated leachate was progressively diluted.

Foliar Ca levels decreased significantly in seedlings irrigated with untreated leachate applied only to the soil and with diluted, carbon-treated leachate. Table 6.8 shows that Ca is removed to some extent by activated carbon, suggesting that further dilution and possibly leaching may cause a Ca deficiency. Elution or displacement and leaching of Ca by other cations in the untreated leachate may also explain the reduced Ca levels in treatments 11-13. Calcium concentrations in treatment 1 indicate that addition of $Ca(OH)_2$ in treatments 5-7 did not cause luxury consumption of this nutrient by the seedlings.

The Cu content of red maple foliage decreased appreciably in plants receiving applications of full and half concentration lime-treated leachate,

presumably due to Cu removal in the precipitated sludge. Manganese was consistently very high across all treatments suggesting that the low soil pH and periodic reducing environment arising from daily irrigation were inducing conditions of high Mn availability in the soil solution. The Mn was thus originating primarily from the local Muskoka Lakes soils and not from the leachate (Table 6.8). Leachate application did not cause any discernable changes in foliar concentrations of P, K, Mg, B or Zn.

Although no visible symptoms of vegetative stress were ever apparent in the adequately but not excessively irrigated red maple seedling trials described above, very different responses were noted in the soil waterlogging trials. Within 24 hours of the initial soil saturation/ drainage treatment cycle (i.e. both 2 and 24 hour duration), sugar maple seedlings exposed to untreated MSW leachate exhibited definite signs of dehydration, permanent (irreversible) wilting and impending mortality. After five such waterlogging/drainage cycles, 100% mortality occurred in the seedlings whose root systems were saturated for 24 hours at a time. Lesser mortality rates were recorded in the seedlings waterlogged for only a 2 hour duration before draining. Seedlings saturated with water for both 2 and 24 hour durations (5 successive treatments), however, showed no signs of reduced vitality.

Table 7.11 presents the results of foliar nutrient analyses for the leaves of the waterlogged seedlings. It should be pointed out that any apparent treatment-related differences in foliar nutrient content must be viewed with caution due to the short duration of this experiment. Nevertheless, some notable trends are evident. Nitrogen originating from the leachate was apparently assimilated at a rapid rate during the five, successive 24 hour leachate treatments, despite the adverse visible effects on the seedlings (i.e. N significantly higher than 24 hour water treatment). Phosphorus was also found limiting in all but the 2 hour water treatment which was undoubtedly the least traumatic of all waterlogging treatments. As one might expect, K uptake was greatest in the short duration leachate treatment (i.e. significantly greater than the 24 hour water treatment) due to increased K availability from the leachate and less pronounced phytotoxic effects as compared to the 24 hour leachate treatment. Calcium uptake tended to decline with prolonged saturation, but this was only statistically significant in the water treatment. Iron uptake was greater in both leachate treatments relative to the 24 hour water treatment. It is believed

Table 7.11. Concentrations of the major and trace nutrient elements in sugar maple seedlings saturated with MSW leachate or water.

Soil Waterlogging Treatment		Foliar Elemental Analysis									
Saturating Liquid	Duration Per Treatment	N	P	K	Ca	Mg	Mn	Fe	B	Zn	Cu
	hr	% $g \cdot g^{-1}$					ppm				
Untreated Leachate	2	2.15 bc	0.05 b	0.73a	0.90 ab	0.20	2275.8	881.9 a	40.13	87.70	71.10
	24	2.47 a	0.08 b	0.57ab	0.75 b	0.18	2375.4	868.5 a	42.19	105.36	72.27
Water	2	2.33 ab	0.46 a	0.67ab	1.05 a	0.21	2538.3	645.9 ab	41.49	86.00	34.09
	24	1.96 c	0.08 b	0.52 b	0.70 b	0.18	1880.7	452.0 b	46.83	109.17	54.17

Means with the same lower case letter are not significantly different at P = 0.05.

that Fe assimilation occurred in the waterlogging experiment and not in the larger greenhouse trials due to the presence of soluble ferrous iron in the reducing soil conditions of the former. No significant patterns of foliar uptake emerged for Mg, B, Zn or Cu, but Mn showed the same very high concentrations due to the chronic reducing conditions maintained in the Muskoka Lakes potting soil.

7.3.1.3 Leachate Osmotic Potential

There are four plausible explanations for the dramatic physiological effects observed in the sugar maple seedlings as a result of cyclic and prolonged waterlogging of the root systems with untreated leachate. The first is a phytotoxic effect of one or more organic or inorganic leachate constituents in direct contact with plant root systems or after being assimilated in plant biomass. The above experimentation suggests that Fe is the most probable of these since plant uptake was evident under waterlogged but not moderately irrigated conditions using untreated leachate. Secondly, the irrigation of coarse-textured, acidic soils will give rise to excessive levels of soluble Mn and possible Mn toxicities. The waterlogging experiment showed somewhat higher foliar Mn levels than the red maple seedling experiment. Thirdly, the relatively high COD and BOD of the leachate (Table 6.2) from soluble and particulate organics will only exacerbate a low soil aeration condition in the plant root zone during soil waterlogging. Soil microbes will rapidly utilize any oxygen in the limited soil air to decompose these organic substances, thus contributing to an anoxic environment, inhibiting root respiration and causing root disfunction. Finally, the osmotic potential of the leachate may be sufficient in an overirrigated or waterlogged soil to create a significant added resistance to water uptake and transpiration by plants. Dissolved solids and solutes in the leachate could also cause plant desiccation if sprayed directly on above-ground shoot vegetation. Of these four possible sources of vegetative stress, only osmotic potential can be readily and definitively evaluated as a factor contributing to forest decline and mortality.

A laboratory investigation was undertaken to estimate the osmotic potential of the leachate at the Muskoka Lakes site. A 1M NaCl solution was prepared using deionized water. According to Van't Hoff's Law, a 1M solution of NaCl has an osmotic potential of -48 bars. A small container of this salt solution was placed in a desiccator with a sample of MSW leachate.

The samples were periodically weighed to ascertain the direction and magnitude of vapour exchange. This procedure was iterated for various concentrations of NaCl in solution. Ideally, when no vapour exchange is observed (i.e. the mass of the two solutions remain constant), the two solutions are in equilibrium and have equivalent osmotic potentials.

The results showed that the osmotic potential of the Muskoka Lakes landfill leachate is approximately -6 bars. The water potential at which vegetation permanently or irreversibly wilts and ultimately dies is about -15 bars. It is, therefore, unlikely that the osmotic potential in itself could induce sufficient desiccation of the plant roots or shoots to have any lasting effect on vegetative vigor.

Water uptake by roots is a function of the total water potential in the soil, or the sum of the matric and osmotic potentials. If dry soil conditions were to lower the matric potential to -9 bars or lower, however, the total water potential would reach the permanent wilting point (-15 bars) and vegetation might experience great difficulty in extracting soil water. Frequent leachate applications to the soil will tend to maintain the soil matric potential between 0 and -1 bar, thus largely eliminating this possibility in the field.

7.4 Conclusions

Both field and laboratory studies presented in this segment of the report provide overwhelming evidence that the principal causative agent (direct and indirect) of vegetative and soil microbial stress at the Muskoka Lakes site is soil waterlogging from excessive leachate application rates. Qualitative understory vegetation distribution and remotely-sensed tree canopy vigour information strongly indicate that daily irrigation as implemented at this site has caused a radical change in the soil moisture regime of the root zone to which the vegetation and soil microbial populations must adapt.

As one might expect, the understory vegetation community has undergone a relatively rapid shift in species distribution with prolonged irrigation in favour of more hydrophyllic shrubs and herbaceous plants. The forest trees are much less adaptable over the short time interval during which spray irrigation has been conducted. If any such transition to tree species better able to withstand wet soil conditions were to occur naturally, it would do so over a much longer period of time. Meanwhile, IR photography shows a general forest decline over much of the current 4.3 ha spray area

and tree mortality in localized areas where excessive loading rates have been applied since 1980 (i.e. spray area 1). The forest transition is thus occurring as a threshold phenomenon with respect to application rates. Only spray area 3 (light spray history) shows few visible ill effects at this point in time.

More quantitative field measurements and a controlled greenhouse seedling experiment provided further confirmation of the chronic soil waterlogging problem. The marked decline in leaf transpiration in an irrigated soil environment is clearly attributable to the creation of poor soil aeration conditions which impede root respiration and function. Increased stomatal resistance in these same leaves will lead to diminished CO_2 fixation and vegetative productivity over the longer term even though stem increment borings show little evidence of reduced annual growth within the last five years in the heavily sprayed area. Tree vegetation was shown to utilize the macronutrient elements present in leachate and certain species (e.g. beech) assimilated soluble leachate metals like Fe and B into their biomass under heavily sprayed conditions. The indigenous vegetation thus exhibited some attenuation capacity and demonstrated that MSW leachates have good potential for recycling in forest ecosystems if properly implemented. The high availability and plant uptake of Mn in these coarse-textured, acidic soils is augmented by irrigation to levels that may well be toxic over the long-term, but these toxicity limits are inadequately defined for most natural species. The greenhouse seedling experiment showed the same high Mn plant tissue levels after harvesting even though no visible damage or toxicity symptoms were in evidence after seven weeks of leachate irrigation. Irrigating native species with leachate at rates consistent with the evapotranspirational demand of the plants has few undesirable short-term effects and precludes Fe uptake due to the predominance of ferric and not ferrous iron under these conditions.

There is evidence that the speciation of the soil microbial population also shifts in response to leachate exposure. Soil microbial biomass was shown to decrease markedly with increasing leachate application rates or longer spray histories but soil respiration was significantly higher within the light spray area relative to the unsprayed control. This apparent dichotomy can only be explained by assuming that the dominant pre-spray microbial species have been displaced by a different microbial population as a result of leachate spraying, possibly from aerobic heterotrophs to Fe-reducing bacteria. This has major implications with respect to nutrient

recycling in the forest ecosystem since it can be assumed that a broad spectrum of microflora such as mycorrhiza (i.e. important in P uptake) will be similarly affected.

Of the four most probable causes of vegetative phytotoxicity with leachate irrigation, only the wastewater osmotic potential has been eliminated from further consideration. Ferrous iron is the most likely major leachate constituent to be phytotoxic when in direct contact with root systems or after being assimilated in plant biomass. Iron is also capable of i) creating subsurface indurated layers in these soils which can cause excess soil water problems due to perched water table formation (section 6.4.4), and ii) directly reducing plant photosynthesis through foliar staining. The relatively low levels of volatile organics in the leachate once it resides from some time in the settling lagoons and is atomized by the spray nozzles are unlikely to be a direct cause of phototoxicity. The presence of recalcitrant, extractable organics was not investigated in this study and cannot be discounted as a possible factor in the observed plant physiological damage. Manganese toxicity arising from the irrigation of coarse-textured, acidic soils is another strong possibility, irrespective of the Mn content of the leachate itself. The high organic content of leachates may indirectly cause a third source of vegetative stress by placing a high demand on available oxygen in the soil air and constraining root respiration as the organics are decomposed in the upper soil zones. All three factors are most likely contributing to observed vegetative and soil microbial stresses since all are linked to soil waterlogging conditions and increase in severity as irrigation creates larger soil moisture surpluses.

8.0 Leachate Pretreatment Prior to Land Disposal

8.1 Preface

This section of the report presents a preliminary assessment of possible means of leachate pretreatment which might lessen the adverse effects of spray irrigation on vegetation and soils at the Muskoka Lakes site. It was recognized before this study began that a necessary precursor to this project phase would be the full resolution of the nature of observed phytotoxic effects on vegetation and the scope of soil physical and chemical property changes with prolonged irrigation. It was also recognized that only the latter could be fully achieved in a one-year study due to the difficulty of isolating the negative plant physiological responses which could be clearly attributed to one or more specific leachate constituents. In spite of these constraints, and with the added support of the Municipality, several pretreatment options which might be feasible to implement at the Muskoka Lakes site were investigated.

Contaminant analyses indicate that toluene and Fe are the organic and inorganic leachate constituents, respectively, of most concern due to their high concentrations and their known or suspected effects on vegetation. For example, Menser (1981) has suggested that the use of landfill leachate as a source of recycled nutrients has little chance of success unless excessive and unbalanced levels of micronutrients (e.g. Fe, Mn) can be regulated. These organic and inorganic contaminants can be removed to differing degrees using techniques as varied as high pressure reverse osmosis, activated carbon filtration, lagoon aeration, slaked lime ($Ca(OH)_2$) addition and filtration through natural materials such as peat moss or tree bark. Some of these methods can be used together in multi-phase pretreatment, such as the Environment Canada mobile RO/activated C unit used for emergency chemical clean-ups. Lime pretreatment of leachate to remove soluble metals, however, has proven incompatible with upflow sludge bed-filtration systems which are designed to lower effluent COD (Kennedy and Guiot, 1986). A full categorization and description of the major leachate treatment methods known can be found in section 2.4 of this report.

8.2 Leachate Pretreatment Possibilities at Muskoka Lakes

8.2.1 Laboratory Procedures

Simple jar tests were performed on samples taken at two different times of the year and from different locations along the treatment/disposal

system. The use of lime to treat landfill leachates has been thoroughly reviewed by Thornton and Blanc (1973) and is likely to be the most simple and cost-effective pretreatment method available given the existing facilities at Muskoka Lakes. Factors quite apart from its effectiveness in soluble metal removal, however, must be considered as well before final adoption.

Leachate was first collected anaerobically in 500 ml stoppered flasks with no headspace from the sump on July 9, 1986. This wastewater was treated in 400 ml lots with a lime slurry (i.e. approximately 17% $g \cdot g^{-1}$) or dry lime to yield final concentrations of 0.5, 1.0, 2.0 and 2.5 g $Ca(OH)_2 \cdot \ell^{-1}$ of leachate as described by Knox (1983). All treatments including a non-limed control were flash stirred (70 rpm) for 5 minutes to achieve rapid mixing, slow stirred (20 rpm) for 20 minutes to allow coagulation and were then allowed to settle for 35 minutes and 24 hours. After each settling time, the supernatant for each treatment was analysed for NH_4^+, P, K, Na, Mg, Mn, Cu, Zn and Fe. The removal of soluble iron from leachate is of particular concern since greenhouse experiments have shown that red maple seedlings irrigated with full-strength (59 ppm Fe) and half-strength (30 ppm Fe) untreated leachate for nine weeks had accumulated mean levels of Fe of 2235 and 1902 ppm, respectively, in their leaves.

A second pretreatment investigation was undertaken in the fall of 1986 involving a much expanded range of pretreatment methods. Leachate was sampled on November 6 from the surface of the settling lagoons and the sump and no attempt was made to limit exposure of these samples to the atmosphere. As above, lime pretreatment was re-investigated on this partially aerated leachate but over a slightly broader range of concentrations (i.e. 0.25 - 2.5 g Ca $(OH)_2 \cdot \ell^{-1}$) and only using the lime slurry. In addition, the effectiveness of 25 minutes of bubbling aeration followed by a 1 hour settling period was evaluated as a pretreatment option which might be carried out in the settling lagoons on site. Other pretreatment methods investigated in this second trial were percolation of the leachate through a column of granular activated carbon (i.e. Calgon Filtrasorb 300) and reverse osmosis as carried out by Zenon Environmental Inc. in Burlington, Ontario. Finally, the ability of various types of tree bark to remove soluble iron from leachate samples was determined using the technique of Vaughan et al. (1984). Briefly, leachate obtained from the settling lagoons was allowed to percolate through a column (70 cm x 30 cm)

of Norway Spruce, White Pine or Red Oak bark fragments about 4 cm^2 in size. The treated leachate from all of the above methods was analysed for NH_4^+, P, K, Mg, Mn, Cu, Zn, Fe, toluene, benzene, ethylbenzene, p- and m-xylene and o-xylene.

8.2.2 Pretreatment Test Results

The analytical results obtained indicate that, of the pretreatment options investigated, granular activated carbon and reverse osmosis provide the highest effluent quality. Table 8.1 provides the treatment performance of the Calgon Filtrasorb 300 for the major organic and inorganic leachate constituents. The untreated organic concentrations for all but toluene are very low as a result of volatilization from the surface of the settling lagoons on site. The pH of the effluent from the activated carbon column was 7.4 relative to the influent value of 5.5.

Table 8.1. Treatment performance of granular activated carbon on leachate sampled from settling lagoons.

Leachate Constituent	Untreated Concentration	Percent Removed by Activated Carbon
	$mg \cdot \ell^{-1}$	%
<u>Organic</u>		
toluene	402	100
o-xylene	2	100
p-, m-xylene	0	-
ethylbenzene	0	-
benzene	0	-
<u>Inorganic</u>		
Fe	59.0	100
Mn	4.1	100
Zn	0.2	100
NH_4^+	40.0	93
P	0.5	60
Ca	117.5	56
Mg	21.4	40
K	49.6	24
Na	30.5	1.3

This escalation in pH undoubtedly contributes to the excellent removal of Fe and Mn through precipitation of the insoluble form of these metals on the high surface area carbon.

Zenon Environmental Inc. in Burlington, Ontario, has been actively working to advance and promote reverse osmosis technology for use in the

clean-up of localized chemical spills and other applications. In doing so, several pilot-scale units have been developed employing different membrane types and configurations. A 100 ℓ volume of the Muskoka Lakes leachate sampled from the culvert well (i.e. highest leachate strength) was sent to Zenon Env. Inc. to ascertain its treatability with several of the existing membranes after ultrafiltration. The results indicated that it was possible to achieve essentially complete removal of both organic and inorganic constituents in the "permeate". A "concentrate" of about 1/10 of the original total volume is created in the process.

In general, lime slurry applications were more effective than dry lime at the same concentrations in removing most of the major inorganic constituents. No appreciable difference in removal efficiency or sludge precipitation was noted between samples allowed to settle for 35 minutes versus 24 hours. The lime slurry method, in turn, performed better on aerobically-sampled (lagoon) than anaerobically-sampled leachate to the point where effectively all Fe and Mn was removed in the precipitating sludge over the full range of lime concentrations. Table 8.2 presents these trends for the 1.0 g $Ca(OH)_2 \cdot \ell^{-1}$ treatment. The lime slurry method also seems to cause a slightly lower increase in pH than dry lime application but the quantity of precipitate or sludge falling out of suspension is greater. Figure 8.1 shows the effect of slaked lime concentration on the efficacy of a lime slurry applied to leachate from the settling lagoons. The lime concentration which optimizes the percent removal of the major inorganic contaminants against the amount of sludge precipitated appears to be about 1.0 g $Ca(OH)_2 \cdot \ell^{-1}$. The increase in precipitate formed with the further addition of lime beyond this concentration would make the marginally higher levels of removal difficult to justify. If the liming treatment were aimed solely at Fe removal, an application rate of 0.25 $g \cdot \ell^{-1}$ would be appropriate for the lagoon leachate. The sump leachate, however, would require 1.0 $g \cdot \ell^{-1}$ to effect 95% Fe removal. As noted by Knox (1983), the murky orange-brown colour of untreated leachate changed to a clear liquid with a grey-green precipitate at the 1.0 $g \cdot \ell^{-1}$ dose in this jar test.

One unexpected finding from the liming tests was the apparent suppression of organic volatilization with increasing $Ca(OH)_2$ dosage. The levels of toluene found in the lime-treated sump leachate were 157, 202, 202, 259 and 358 $\mu g \cdot \ell^{-1}$ for $Ca(OH)_2$ slurry concentrations of 0.25, 0.50, 1.0, 2.0 and 2.5 $g \cdot \ell^{-1}$ of leachate. The untreated sump leachate control contained 402 $\mu g \cdot \ell^{-1}$ of toluene. This trend was not found with the lime-

Table 8.2. Treatment of aerobically- and anaerobically-sampled leachates by addition of lime slurry and dry lime to a concentration of 1.0 g $Ca(OH)_2 \cdot \ell^{-1}$ leachate.

Chemical Parameter	Aerobically-Sampled Leachate (Lagoon)			Anaerobically-Sampled Leachate (Sump)				
	Control $mg \cdot \ell^{-1}$	Lime Slurry $mg \cdot \ell^{-1}$	Removal %	Control $mg \cdot \ell^{-1}$	Lime Slurry $mg \cdot \ell^{-1}$	Removal %	Dry Lime $mg \cdot \ell^{-1}$	Removal %
Fe	59.00	0.0	100.0	41.00	2.00	95.1	20.00	51.2
Mn	4.05	0.0	100.0	7.10	0.70	90.1	0.90	87.3
Mg	21.40	0.62	97.1	59.00	63.00	- 6.8	44.00	25.4
P	0.50	0.08	84.0	0.15	0.20	- 33.3	0.20	- 33.3
NH_4^+	40.00	20.50	48.8	67.00	57.00	14.9	55.00	17.9
K	49.60	39.60	20.2	90.00	92.00	- 2.2	86.00	4.4
Na	30.50	25.70	15.7	70.00	67.00	4.3	72.00	- 2.9
Ca	117.50	201.00	-71.1	n.d.	n.d.	n.d.	n.d.	n.d.
Precipitate (cm)	n.d.	n.d.	n.d.	0.20	1.35	-575.0	0.70	-250.0
pH	5.50	7.70	-	5.47	7.49	-	7.88	-

n.d. = not determined

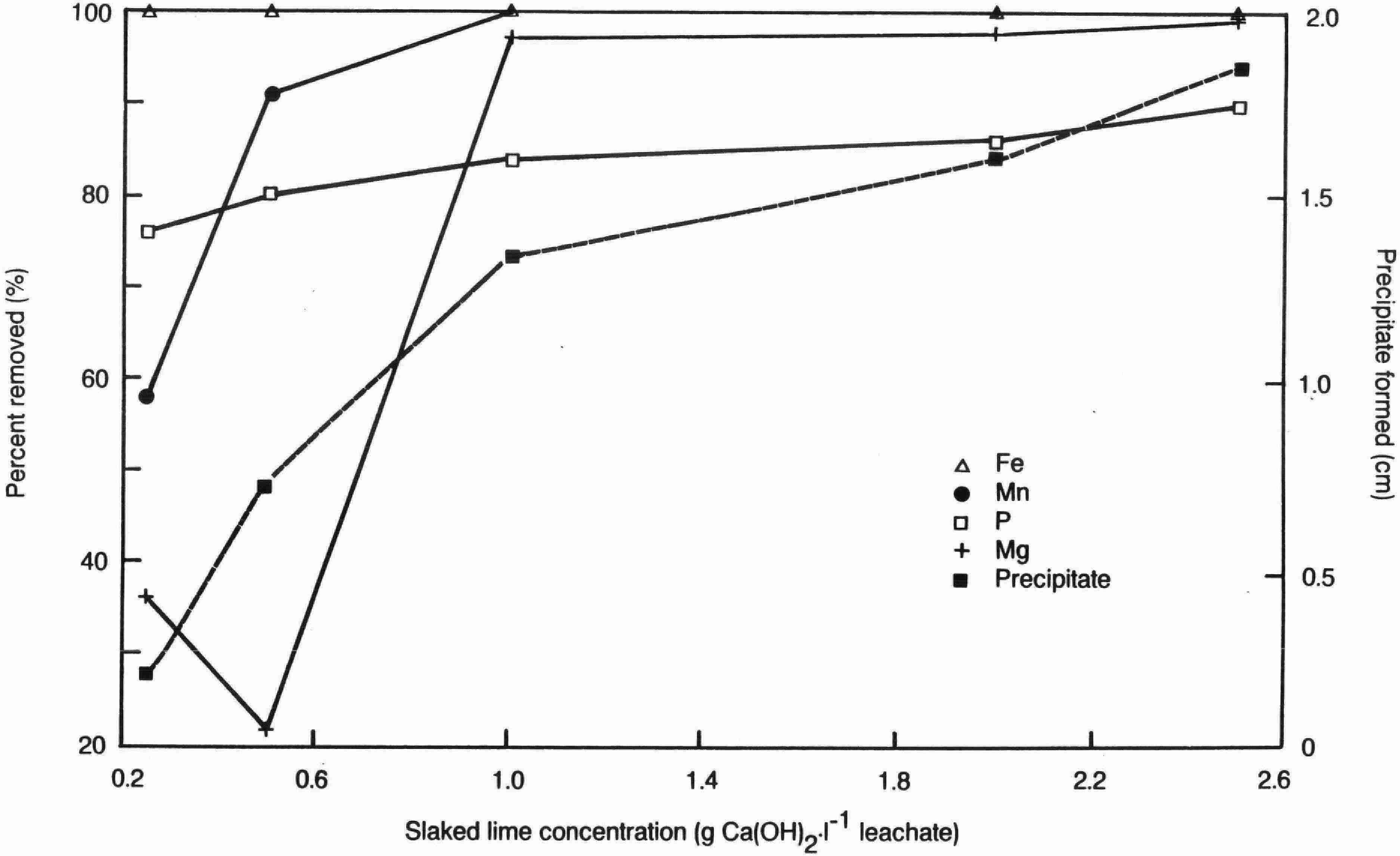


Figure 8.1. Treatment performance and precipitate formation by addition of lime slurry to lagoon-sampled leachate.

treated lagoon leachate, but the toluene concentration was much lower at the outset (i.e. 4 $\mu g \cdot \ell^{-1}$). The mechanism causing this suppression of volatilization is not known but this finding further underscores the need to minimize lime application rates if implemented as a pretreatment method.

Aeration tests (i.e. 25 minute bubbling aeration) with sump and lagoon leachates determined that further forced aeration of the lagoon leachate did not result in substantially improved organic or inorganic constituent removal. The pre-spray removal efficiency already obtained within the limited aerobic environment of the settling lagoons themselves is considerable (Table 8.3). This observation, however, must be viewed with some caution since the lagoon samples were taken from the very surface and may not be entirely representative of the lagoon leachate at depth. Quiescent ponding has effected Fe, P and toluene removal of 96%, 80% and 99%, respectively, with 21% to 34% of the remaining constituents being removed. Forced aeration also proved most effective for the removal of Fe, P and toluene in the sump leachate (Table 8.3). Only Zn showed no removal in either the aerated or untreated lagoon leachate. The solution pH tended to increase dramatically with forced aeration relative to the untreated sump and lagoon leachate samples. Bubbling aeration of the sump leachate for a short duration thus does not appear to be as effective as more prolonged, quiescent retention in the lagoons.

Highly weathered conifer tree bark has been used successfully in the U.K. to remove soluble ferrous iron from tile drainage water to help prevent ochre accumulation at the point of tile outflow (section 2.4.3.3). Vaughan et al. (1984) found that after only one minute of shaking, Norway Spruce and Scotch Pine bark had absorbed 60-80% of the ferrous iron from solutions containing up to 400 mg $Fe \cdot \ell^{-1}$ and that this increased to 85% removal after a 10 minute exposure to the bark. Attempts to reproduce these findings using leachate taken from the settling lagoons and sump were not sufficiently successful to base any recommendations on at this time. The degree of bark weathering, size of fragments used, column packing density and leachate retention time are all factors which could account for the lesser removal efficiencies found in this study compared with those of Vaughan et al. (1984). It is more likely, however, that the limited degree of aeration which occurs in the settling lagoons is sufficient to alter the redox state of the iron and preclude its absorption in large quantities by the bark. Further lab experimentation is warranted on this natural treatment medium using anaerobically-sampled leachate from the sump or culvert well.

Table 8.3. Treatment performance of 25 minutes of bubbling aeration on sump and lagoon leachates.

Leachate Constituent	Sump Untreated	% Removed Relative to Untreated Sump		
		Lagoon Untreated	Lagoon Aerated	Sump Aerated
	$mg \cdot \ell^{-1}$	%	%	%
Fe	59.00	95.9	95.2	60.1
P	0.50	80.0	80.0	80.0
Mg	21.40	33.6	35.9	5.1
Mn	4.05	33.0	31.3	- 0.7
K	49.60	32.0	32.0	9.6
Ca	117.50	31.4	33.6	5.1
NH_4^+	40.00	25.0	20.0	- 1.2
Na	30.50	20.9	27.5	10.1
Zn	0.20	0.0	0.0	-50.0
toluene	0.402	99.0	100.0	62.7
pH	5.50	6.30	7.50	7.50

8.3 Conclusions

The process of "balancing" or quiescent settling that is now taking place in the double lagoons on site would appear to be effecting a reasonable level of aerobic pretreatment on the leachate before it is distributed into the forest. This is particularly true for the volatile organics which show little if any difference in concentration between samples taken from the very surface of the lagoon and those from spray nozzle emissions. Even though Fe levels decrease significantly in the leachate taken from the lagoon surface due to coagulation and precipitation, all soluble metals entering the lagoons must eventually be distributed within the spray areas since no attempt is now made to remove any sludge accumulating on the lagoon bottoms. Laboratory bubbling aeration tests would suggest that forced aeration of the lagoons would not effect appreciably greater precipitation of inorganic constituents including Fe.

Lime pretreatment induces much more rapid Fe precipitation than aeration through sludge formation and sedimentation. The application of a slaked lime slurry (17% $g \cdot g^{-1}$) at a concentration of 1.0 g $Ca(OH)_2 \cdot \ell^{-1}$ of leachate produces a highly-treated, clear effluent but large quantities of a grey-green sludge. At current leachate generation rates, this implies that

there would be at least 20 tonnes of sludge on a dry weight basis to be disposed of each year if lime pretreatment were adopted. A complete characterization of the contaminants in the sludge and their propensity for accumulating in plant biomass would have to be carried out periodically if the sludge were to be used in lieu of a lime soil amendment on local agricultural land. Other considerations include i) the ease with which this sludge could be removed from the existing lagoons which were not designed for this purpose, ii) the increased likelihood of precipitate formation in the spray distribution network and ensuing maintenance problems and costs, iii) the apparent suppression of organic contaminant volatilization with lime application, and iv) the even more pronounced increase in soil pH than currently being observed in the forested spray area with prolonged spraying of lime-treated effluent.

Lime addition can cause large increases in pH in soils of low buffering capacity which may greatly influence nutrient availability and toxicity. Spraying untreated leachate first induces a major increase in soil pH followed by the formation of a podzolic B horizon with prolonged loading of Fe-laden wastewater. The soil taxonomy thus changes from a Dystric or Sombric Brunisol (pH < 5.5) to a Melanic or Eutric Brunisol (pH > 5.5) and finally to a Ferro-Humic Podzol. Dramatic changes in the nutrient regime due to this transition could place major stresses on the indigenous vegetation and contribute to forest decline.

The most inconclusive results were from tree bark pretreatment where clearly more experimentation is needed to fully establish the applicability of this potential low-cost method of Fe removal to landfill leachates. Reverse osmosis is the most promising pretreatment method investigated since the "permeate" would be of high enough quality to discharge directly to the perennial stream while the "concentrate" could be recycled through the landfill and/or irrigated in the forested spray area. The cost of a package RO unit has been estimated to be well below \$100,000, excluding the cost of a full-time operator (H. Donison, pers. comm.) This compares favourably to the cost of a package activated sludge reactor capable of handling about $4.5 \times 10^4 \ell \cdot d^{-1}$ which is closer to \$1/4 M. Activated carbon can effect wide spectrum removal of both organic and inorganic contaminants but a pretreatment facility based on carbon filtration alone would be

---

Heather Donison, Zenon Environmental Inc., 845 Harrington Court, Burlington, Ontario

prohibitively expensive due to carbon replacement and/or re-activation costs. Coupled with a RO unit, activated carbon would be better used as a final permeate polishing step before discharge to the watercourse.

All of the above conclusions are based on laboratory experimentation. Practical experience frequently shows that treatment efficiencies obtained within the confines of a lab are superior to those found in plant-scale facilities in the field which are based on the same processes. This section of the report outlines the findings for a broad range of pretreatment possibilities and thus forms a basis for further work where appreciable resources are dedicated solely to this type of experimentation. Any attempt to implement any of the above forms of pretreatment must be preceded by more detailed laboratory study geared toward the establishment of engineering design criteria.

9.0 Literature Cited

Anderson, J.M. 1973. Carbon dioxide evolution from two temperate deciduous woodland soils. J. Appl. Ecol. 10: 361-378.

Anonymous. 1985. Silvicycle: Information network of waste utilization in forestlands. Institute of Forest Resources, University of Washington. Vol. 3.

Applegate, L.E. 1984. Membrane separation processes. Chem. Eng. 11: 64-89.

Barber, C. and P.J. Maris. 1984. Recirculation of leachate as a landfill management option: benefits and operational problems. Q.J. Eng. Geol. (London) 17: 19-29.

Beadle, C.L., M.M. Ludlow, and J.L. Honeysett. 1985. Water relations. In Techniques in Bioproductivity and Photosynthesis. J. Coombs, D.O. Hall, S.P. Long, and J.M.O. Scurlock (Eds.) 2nd Ed. Pergamen Press: p. 50-54.

Bennett, O.L., H.A. Menser and W.M. Winant. 1975. Land disposal of leachate from a municipal sanitary landfill. Proc. 2nd Natl. Conf. on Compl. Wat. Reuse. A.I.C.E. and E.P.A. Tech. Trans. p. 789-800.

Boyle, W.C. and R.K. Ham. 1974. Biological treatability of landfill leachate. J. Water Poll. Control. Fed. 46(5): 860-872.

Brookes, P.C., J.F. Kragt, D.S. Powlson and D.S. Jenkinson. 1985. Chloroform fumigation and the release of soil nitrogen: The effects of fumigation time and temperature. Soil Bio. Biochem. 17: 831-835.

Brookes, P.C., D.S. Powlson and D.S. Jenkinson. 1984. Phosphorus in the soil microbial biomass. Soil Bio. Biochem. 16: 169-175.

Canada Soil Survey Committee, Subcommittee on Soil Classification. 1978. The Canadian system of soil classification. Can. Dept. Agric. Publ. 1646. Supply and Services Canada, Ottawa, Ont. 164 pp.

Cartwright, P. 1985. Reverse osmosis. Water and Waste Treatment 28(3): 24-27.

Chian, E.S.K. and F.B. Dewalle. 1976. Sanitary landfill leachates and their treatment. J. Env. Eng. A.S.C.E. 102: 411-431.

Clarholm, M. and T. Rosswall. 1980. Biomass and turnover of bacteria in a forest soil and a peat. Soil Bio. Biochem. 12: 49-57.

Cold Regions Research and Engineering Laboratory. 1984. Impact of slow rate land treatment on groundwater quality. U.S. Army Corps. of Engineers, CRREL. L.V. Parker, T.F. Jenkins and B.T. Foley (eds.). USEPA Report. Hanover, NH.

Cook, E.N. and E.G. Foree. 1974. Aerobic biostabilization of sanitary landfill leachate. J. Water Poll. Control Fed. 46(2): 380-392.

DeKimpe, C.R., M.R. Laverdiere and J. Dejou. 1983. Distribution of silica, sesquioxides and clay in Quebec podzolic soils and their effects on subsoil cementation. Soil Sci. Soc. Am. J. 47: 838-840.

Dewalle, F.B. and E.S.K. Chian. 1974. Removal of organic matter by activated carbon columns. J. Env. Eng. A.S.C.E. 100: 1089-1104.

Dore, W.G. and J. McNeill. 1980. Grasses of Ontario. Monograph 26. Research Branch, Agriculture Canada, Ottawa, Ontario.

Edwards, N.T. 1982. The use of soda lime for measuring respiration rates in terrestrial systems. Pedobiologia 23: 321-330.

Ferderle, T.W., D.C. Dobbins, J.R. Thornton-Manning and D.D. Jones. 1986. Microbial biomass, activity, and community structure in subsurface soils. Groundwater 24: 365-374.

Fernald, M.L. 1970. Gray's Manual of Botany. Eighth Edition. D. Van Nosterand Company, New York.

Foy, C.D., R.L. Chaney and M.D. White. 1978. The physiology of metal toxicity in plants. Ann. Rev. Plant Physiol. 29: 511-566.

Fraser, J. and A. Sims. 1983. Tip leachate treatment using hydrogen peroxide. Wat. Poll. Control 82(2): 243-245.

Freeze, R.A. and J.A. Cherry. 1979. Groundwater. Prentice-Hall Inc., N.J.

Gartner Lee Associates Ltd. 1977. Preliminary hydrogeological study of the Township of Muskoka Lakes landfill. 7 pp.

Gartner Lee Associates Ltd. 1978. Township of Muskoka Lakes landfill leachate effects and treatment options. 16 pp.

Gartner Lee Associates Ltd. 1984. Soil investigation for proposed leachate spray irrigation system - Pinelands landfill site. Final Report. 20 pp.

Gee, G.W. and C.A. Federer. 1972. Stomatal resistance during senescence of hardwood leaves. Water Resources Res. 8: 1456-1460.

Gersberg, R.M., B.V. Elkins, S.R. Lyon and C.R. Goldman. 1986. Role of aquatic plants in wastewater treatment by artificial wetlands. Water Res. 20(3): 363-368.

Gleason, H.A. and A. Cronquist. 1963. Manual of Vascular Plants of Northeastern United States and Adjacent Canada. William Grant Press, Boston, Mass.

Gordon, A.M., R.E. Schlenter and K. Van Cleve. In press. Seasonal patterns of soil respiration and CO_2 evolution following harvesting in the white spruce forests of interior Alaska. Can. J. For. Res.

Greenhouse, J.P. and R.D. Harris. 1983. Migration of contaminants in groundwater at a landfill: A case study VII. Direct current, very low frequency and inductive resistivity surveys. J. of Hydrology 63: 177-197.

Griffin, R.A. et al. 1976. Attenuation of pollutants in municipal landfill leachate by clay minerals: Column leaching and field verification. Env. Geology Notes, No. 78. Illinois State Geological Survey, Urbana, IL. 34 pp.

Grover, B.L. and R.E. Lamborn. 1970. Preparation of porous ceramic cups to be used for extraction of soil water having low solute concentrations. Soil Sci. Soc. Am. Proc. 34(4): 706-708.

Hansen, E.A. and A.R. Harris. 1975. Validity of soil-water samples collected with porous ceramic cups. Soil Sci. Soc. Am. Proc. 39: 528-536.

Hantzsche, N.N. 1985. Wetland systems for wastewater treatment: Engineering applications. In Ecological Considerations in Wetlands Treatment of Municipal Wastewaters. P.J. Godfrey, E.R. Taylor, S. Pelzarski, J. Benforado (Eds.). Van Nostrand Reinhold Co. Inc., New York. p. 7-25.

Harrington, D.W. and P.J. Maris. 1986. The treatment of leachate: A U.K. perspective. Water Poll. Control 85: 45-56.

Harris, A.R. 1978. Evaluation of several methods of applying sewage effluent to forested soils in the winter. U.S.D.A. Forest Service, North Central Expt. Station Res. Paper NC-162. 8 pp.

Ho, S., W.C. Boyle and R.K. Ham. 1974. Chemical treatment of leachates from sanitary landfills. J. Water Poll. Control Fed. 46(7): 1776-1791.

Iskander, I.K. 1981. Modeling wastewater renovation - land treatment. John Wiley and Sons, New York, N.Y.

Jenkinson, D.S. and D.S. Powlson. 1976. The effects of biocidal treatments on metabolism in soil. V. A method for measuring soil biomass. Soil Bio. Biochem. 8: 209-213.

Jurik, T.W. 1986. Seasonal patterns of leaf photosynthetic capacity in successional northern hardwood tree species. Amer. J. Bot. 73(1): 131-138.

Keenan, J.D., R.L. Steiner and A.A. Fungaroli. 1984. Landfill leachate treatment. J. Water Poll. Control Fed. 56(1): 27-33.

Kennedy, K.J. and S.R. Guiot. 1986. Anaerobic upflow bed-filter: Development and application. Proc. of Symp. on Anaerobic Fixed-Film Digestion, Toronto, Ontario. p. 15-30.

Klefstad, G., L.V. Sendlein and R.C. Palmquist. 1975. Limitations of the electrical resistivity measurements in landfill investigations. Groundwater 13(5): 418-427.

Knox, K. 1983. Treatability studies on leachate from a co-disposal landfill. Env. Poll. Series B. 5: 157-174.

Lancashire County Council. 1984. Treatment of leachate. County Survey Society Committee #4. Special Activity Group #6. p. 32-61.

Lee, G.F., R.A. Jones and C. Ray. 1986. Sanitary landfill leachate recycle. Biocycle. 27(1): 36-38.

Levin, M.J. and D.R. Jackson. 1977. A comparison of in situ extractors for sampling soil water. Soil Sci. Soc. Am. J. 41: 535-536.

LI-COR Inc. 1984. Steady State Porometer: LI-1600. Instruction Manual. Pub. No. 8107-01R3. Lincoln, Nebraska.

Lozano, F.C. and I.K. Morrison. 1981. Distribution of hardwood nutrition by sulphur dioxide, nickel and copper air pollution near Sudbury, Canada. J. Env. Qual. 10: 198-200.

MacFarlane, D.S., J.A. Cherry, R.W. Gillham and E.A. Sudicky. 1983. Migration of contaminants in groundwater at a landfill: A case study I. Groundwater flow and plume delineation. J. of Hydrology 63: 1-29.

Maye, P.R. 1972. Landfill stabilization with leachate recirculation. Georgia Inst. of Tech., School of Civ. Eng., Atlanta, Georgia.

Menser, H.A. 1981. Irrigating with landfill leachate. Biocycle 22(2): 39-41.

Menser, H.A., W.M. Winant, O.L. Bennett and P.E. Lundberg. 1978. Decontamination of leachate from a sanitary landfill by spray-irrigation of a forested ecosystem. Proc. First Annual Conf. of Applied Research and Practice on Municipal and Industrial Waste.

Menser, H.A., W.M. Winant and O.L. Bennett. 1979. Spray irrigation: A land disposal practice for decontaminating leachate from sanitary landfills. U.S.D.A. Report ARR-NE-4. West Virginia University.

Menser, H.A., W.M. Winant and O.L. Bennett. 1983. Spray irrigation with landfill leachate. Biocycle 24(3): 22-25.

M.M. Dillon Ltd. 1980. Township of Muskoka Lakes Pinelands landfill. 13 pp.

Miller, R.H. 1979. The soil as a biological filter. In Utilization of Municipal Sewage Effluent and Sludge on Forest and Disturbed Land. W.E. Sopper and S.N. Kerr (eds.). Pennsylvania State Univ. Press, University Park. p. 71-94.

Moore, I.D., G.J. Burch and P.J. Wallbrink. 1986. Preferential flow and hydraulic conductivity of forest soils. Soil Sci. Soc. Am. J. 50(4): 876-881.

Mueller-Dombois, D. and H. Ellenberg. 1974. Aims and methods of vegetation ecology. John Wiley and Sons, New York, N.Y.

Murtha, P.A. 1972. A guide to air photo interpretation of forest damage in Canada. Canadian Forestry Service, Publication No. 1292, Ottawa, Ont. 63 pp.

Newton, J. 1979. Leachate treatment on landfill sites. Surveyor (London) 19: 33-35.

Nordgren, A., T. Kauri, E. Baath and B. Soderstrom. 1986. Soil microbial activity, mycelial lengths and physiological groups of bacteria in a heavy metal polluted area. Env. Pollution 41: 89-100.

Nordstedt, R.A., L.B. Baldwin and L.M. Rhodes. 1975. Land disposal of effluent from a sanitary landfill. J. Water Poll. Control Fed. 47(7): 1961-1970.

Ontario Ministry of the Environment. 1975. Hydrogeologic survey of proposed solid waste disposal site. Township of Muskoka Lakes, Medora Ward (Pinelands Site). 32 pp.

Oron, G., J. DeMalach and J.E. Bearman. 1986. Trickle irrigation of wheat applying renovated wastewater. Water Resources Bulletin 22(3): 439-446.

Pohland, F.G. 1980. Leachate recycle as a landfill management option. J. Env. Eng. A.S.C.E. 106: 1057-1069.

Prouty, M.F. 1986. Upgrading with spray irrigation. Biocycle 27(2): 54.

Rak, S.F. 1984. Reverse osmosis membrane fouling and pretreatment considerations. Industrial Water Engineering 21(4): 12-15.

Reynolds, W.D., D.E. Elrick and G.C. Topp. 1983. A re-examination of the constant head well permeameter method for measuring saturated hydraulic conductivity above the water table. Soil Science 136: 250-268.

Richardson, C.J. and D.S. Nichols. 1985. Ecological analysis of wastewater management criteria in wetland ecosystems. In Ecological Considerations in Wetlands Treatment of Municipal Wastewaters. P.J. Godfrey, E.R. Taylor, S. Pelzarski and J. Benforado (Eds.). Van Nostrand Reinhold Co. Inc., New York. p. 351-392.

Ricklefs, R.E. and K.K. Matthew. 1982. Chemical characteristics of the foliage of some deciduous trees in southeastern Ontario. Can. J. Bot. 60: 2037-2045.

Robinson, H.D and J.L. Lucas. 1985. Leachate attenuation in the unsaturated zone beneath landfills: Instrumentation and monitoring of a site in Southern England. Water Science Tech. 17: 477-492.

Robinson, H.D. and P.J. Maris. 1979. Leachate from domestic waste: Generation, composition and treatment. A review. Water Research Centre, Tech. Report TR108. 38 pp.

Robinson, H.D. and P.J. Maris. 1985. The treatment of leachates from domestic wastes in landfill sites. J. Water Poll. Control. Fed. 57(1): 30-38.

Rowe, A. 1979. Tip leachate treatment by land irrigation. Solid Wastes (London): 603-623.

Salonius, P.O. and M.K Mahendrappa. 1979. Respiration and nitrogen immobilization in forest soil treated with sulfur and urea. Soil Science 127: 358-362.

Schauer, D. 1986. Spray irrigation challenged in Vermont. Biocycle 27(6): 52-54.

Schlenter, R.E. and K. Van Cleve. 1985. Relationships between CO_2 evolution from soil, substrate temperature and substrate moisture in four mature forest types in interior Alaska. Can. J. For. Res. 15: 97-107.

Schomaker, N. 1986. An outline of US-EPA research on land disposal of hazardous waste. Proc. Tech. Trans. Conf., Toronto, Canada. Dec. 8-9. p. 26-37.

Shaw, M.G. 1985. Pretreatment for RO systems. Water and Waste Treatment 28(11): 20.

Singh, J.S. and S.R. Gupta. 1977. Plant decomposition and soil respiration in terrestrial ecosystems. Bot. Rev. 43: 449-528.

Skujins, J., H.O. Nohrstedt and S. Oden. 1986. Development of a sensitive biological method for the determination of a low-level toxic contamination in soils. Swedish J. Agric. Res. 16: 113-116.

Soper, J.H. and M.L. Heimburger. 1982. Shrubs of Ontario. Royal Ontario Museum, Toronto.

Sopper, W.E. and L.T. Kardos. 1979. Vegetation responses to irrigation with treated municipal wastewater In Utilization of Municipal Sewage Effluent on Forest and Disturbed Land. (ed. W.E. Sopper and S.N. Kerr). Penn State Univ. Press, University Park, Pennsylvania. pp. 573.

Straub, W.A. and D.R. Lynch. 1982a. Models of landfill leaching: Moisture flow and inorganic strength. J. Env. Eng. A.S.C.E. 108: 231-250.

Straub, W.A. and D.R. Lynch. 1982b. Models of landfill leaching: Organic strength. J. Env. Eng. A.S.C.E. 108: 251-268.

Thornton, R.J. and F.C. Blanc. 1973. Leachate treatment by coagulation and precipitation. J. Env. Eng. A.S.C.E. 99: 535-544.

Tilton, D.L. and R.H. Kadlec. 1979. The utilization of a fresh-water wetland for nutrient removal from secondarily treated wastewater effluent. J. Env. Qual. 8(3): 328-334.

Tittlebaum, M.E. 1982. Organic carbon content stabilization through landfill leachate recirculation. J. Water Poll. Control Fed. 54(4): 428-433.

Totten, Sims, Hubicki Associates Ltd. 1983. Township of Muskoka Lakes leachate treatment study for Pinelands landfill site. 11 pp.

Turner, J., S.F. Dice, D.W. Cole and S.P. Gressel. 1978. Variation of nutrients in forest tree foliage. A review.

Turner, N.C. 1969. Stomatal resistance to transpiration in three contrasting canopies. Crop Science 9: 303-307.

Uloth, V.C. and D.S. Mavinic. 1977. Aerobic biotreatment of a high-strength leachate. J. Env. Eng. A.S.C.E. 103: 647-661.

United States Environmental Protection Agency. 1980. Lining of waste impoundment and disposal facilities. Office of Water and Waste Management, Washington, D.C. 385 pp.

van Beers, W.F.J. 1963. The auger-hole method: A field measurement of the hydraulic conductivity of soil below the water table. International Institute for Land Reclamation and Improvement. Bulletin 1. Wageningen, The Netherlands. 32 pp.

Vaughan, D., R.E. Wheatley and B.G. Ord. 1984. Removal of ferrous iron from field drainage waters by conifer bark. J. Soil Sci. 35: 149-153.

Voss, E.G. 1972. Michigan Flora Part I: Gymnosperms and monocots. Cranbrook Institute of Science. Bulletin 55, Bloomfield Hills, Michigan.

Voss, E.G. 1985. Michigan Flora Part II: Dicots (Saururacae - Cornaceae). Cranbrook Institute of Science. Bulletin 59, Ann Arbor, Michigan.

Wanjura, D.F. and J.L. Hatfield. 1986. PAR and IR reflectance, transmittance and absorptance of four crop canopies. Trans. A.S.A.E. 29(1): 143-150.

Weber, M.G. 1985. Forest soil respiration in eastern Ontario jack pine ecosystems. Can. J. For. Res. 15: 1069-1075.

Wile, I., G. Miller and S. Black. 1985. Design and use of artificial wetlands. In Ecological Considerations in Wetlands Treatment of Municipal Wastewaters. P.J. Godfrey, E.R. Taylor, S. Pelzarski and J. Benforado (Eds.). Van Nostrand Reinhold Co. Inc., New York. p. 26-37.

Zirschky, J., A.R. Abernathy and J. Braswell. 1986. Land application of wastewater. Water Pollution Control Federation Journal 58(6): 532-534.

APPENDICES

Appendix A

Review Article by Zirschky et al. (1986) on Land Application
of Wastewater

Land application of wastewater

John Zirschky, A. Ray Abernathy, Jennifer Braswell

Loehr and Overcash[1] described the limiting constituent approach to the design of land treatment systems and presented four case studies. A review of groundwater monitoring requirements for municipal wastewater land treatment systems was presented by Zirschky *et al.*[2] Kowal[3] discussed concentrations, transport mechanisms, and routes of exposure of organics, trace elements, sodium, and nitrate during land treatment.

SLOW RATE

Pettygrove and Asano[4] edited a guidance manual for wastewater irrigation. Field investigations and computer simulations conducted for design of a combination rapid infiltration and slow rate system were discussed by Donovan and Evans.[5] Parker *et al.*[6] investigated the removal of 16 toxic organics in slow rate land treatment. Removal efficiency based on percolate concentration, exceeded 98% for all compounds studied. Only chloroform was continuously detected in the percolate. Volatilization was a primary removal mechanism. An extensive review of the fate of trace organics during municipal wastewater land application was presented by Hutchins *et al.*[7] Discussions of processes that affect the fate of organics, groundwater transport, and case studies were included.

The use of reed canarygrass for spray irrigation was discussed by Zeiders and Sherwood.[8] Nitrate-nitrogen was found to accumulate to potentially unsafe levels for livestock in both reed canarygrass and alfalfa irrigated with rendering plant wastewater. Reed canarygrass yields were doubled by application of 10 cm/a, while alfalfa yield did not increase over irrigation water and fertilizer. Plant uptake was approximately 30% of applied nitrogen.[9] Sorghum was found to grow taller and produce a higher grain and forage yield when irrigated with municipal wastewater.[10]

Nitrate concentration in coastal bermudagrass irrigated with swine lagoon effluent approached, but did not exceed, concentrations unsafe for livestock at nitrogen loading rates of 335, 670, and 1340 kg N/ha·a.[11] At loading rates of 670 and 1340 kg N/ha·h, groundwater contamination by nitrates could occur.[12] Nitrogen and phosphorus concentrations in runoff from swine lagoon effluent irrigation areas averaged 7 to 13 mg/L and 3 to 5 mg/L, respectively. Lagoon effluent was not applied, however, during the time of year when runoff was highest.[13] Nagpal[14] found that 60 to 90% of effluent phosphorus was retained in the upper 60 cm of the soil column and that phosphorus retention depended primarily on the effluent percolation rate and leaching period.

OVERLAND FLOW

Thomas[15] reviewed advances in knowledge regarding the overland flow process. Smith and Schroeder[16] presented a model and family of curves for design of overland flow treatment systems with design and operating guidelines. Abernathy *et al.*[17] partially validated the model presented by Smith and Schroeder[16] and discussed the operating experience of the Easley, S. C., system.

Overland flow provided good year-round treatment of both primary and secondary wastewater in Florida.[18] In the cold climate of Laramie, Wyo., however, overland flow produced secondary quality effluent from May through October only.[19] Removal of 13 trace organic compounds added to wastewater was studied by Jenkins *et al.*,[20] using a prototype overland flow system. Greater than 90% removal was found for each compound, and sorption on the soil followed by biodegradation or volatilization were suggested as the removal mechanisms.

RAPID INFILTRATION

Crites[21] reviewed nitrogen and phosphorus removal efficiencies at several systems. Sustained nitrogen removals from primary effluent of 75% can be obtained by optimizing the dosing schedule.[22] Reed *et al.*[23] discussed the factors leading to problems at several rapid infiltration systems. Poor field investigation and construction procedures were primary factors. Predictions of phosphate movement through a soil column using a mathematical model for phosphorus transport were compared with laboratory results and determined to be inadequate by Mansell *et al.*[24] because of laboratory procedures used to obtain reaction rate coefficients. A finite-difference model for predicting phosphorus transport and sorption in rapid infiltration systems was presented by Mansell *et al.*[25]

MISCELLANEOUS

Effects of construction practices and operational strategies on soil hydraulic properties were investigated by Tyler *et al.*[26] Heavy machinery can significantly reduce the soil infiltration rate and removal of the compacted soil layers can restore infiltration capacity. The use of color infrared aerial surveys for evaluating soil absorption systems including the procedures and equipment for such surveys was discussed by Farrell.[27] Weber and Scott[28] investigated ammonia removal in water hyacinth systems and concluded that ammonia removal was a function of hydraulic application rate and reactor length and independent of the ammonia loading rate. Ecological considerations associated with wetlands treatment of wastewater are discussed in a symposium proceedings edited by Godfrey, *et al.*[29]

John Zirschky and Jennifer Braswell are environmental engineers with ERM-Southeast, Inc., and A. Ray Abernathy is a professor in the Environmental Systems Engineering Department at Clemson University, Clemson, South Carolina. Correspondence should be addressed to John Zirschky ERM-Southeast, Inc. Suite 201, 2623 Sandy Plains Road, Marietta, GA 30066.

REFERENCES

1. Loehr, R. C., and Overcash, M. R., "Land Treatment of Wastes: Concepts and General Design." *J. Environ. Eng. Div., Proc. Am. Soc. Civ. Eng.*, **111**, 141 (1985).
2. Zirschky, J., *et al.*, "Ground Water Monitoring at Land Treatment Sites." *Biocycle*, **26**, 50 (1985).
3. Kowall, N. E., "Health Effects of Land Treatment: Toxicology." EPA-600/1-84-030, NTIS PB85-178473, U. S. EPA, Cincinnati, Ohio (1985).
4. Pettygrove, G. S., and Asano, T., (Eds.), "Irrigation with Reclaimed Municipal Wastewater—A Guidance Manual." Lewis Publishers, Chelsea, Mich. (1985).
5. Donovan, J. F., and Evans, R. R., "Groundwater Studies for Land Treatment Facility." *Proc. 1985 Natl. Environ. Eng. Conf., Amer. Soc. Civ. Eng.*, J. C. O'Shaughessy (Ed.), Amer. Soc. Civ. Eng., New York, N. Y., 144 (1985).
6. Parker, L. V., *et al.*, "Impact of Slow-Rate Land Treatment on Groundwater Quality, Toxic Organics." CRREL-84-30, U. S. Army Cold Regions Research and Engineering Laboratory, Hanover, N. H. (1984).
7. Huchins, S. R., *et al.*, "Fate of Trace Organics during Land Application of Municipal Wastewater." *CRC Rev. Environ. Control* **15**, 355 (1985).
8. Zeiders, K. E., and Sherwood, R. T., "Effects of Sprinkler Irrigation with Municipal Sewage Wastewater and Cutting Management on Nutritive Value, Growth, and Disease Severity of Reed Canary Grass." USDA/ARS-22, PB 85-204744, Agricultural Research Service, Beltsville, Md., (1985).
9. Bole, J. B., and Gould, W. D., "Irrigation of Forages with Rendering Plant Wastewater: Forage Yield and Nitrogen Dynamics." *J. Environ. Qual.*, **14**, 119 (1985).
10. Day, A. D., and Cluff, C. B., "Municipal Wastewater Increases Crop Yields." *Biocycle*, **16**, 48 (1985).
11. Burns, J. C., "Swine Lagoon Effluent Applied to 'Coastal' Bermuda Grass: I. Forage Yield, Quality, and Element Removal." *J. Environ. Qual.*, **14**, 9 (1985).
12. King, L. D., *et al.*, "Swine Lagoon Effluent Applied to 'Coastal' Bermuda Grass. II. Effects on Soil." *J. Environ. Qual.*, **14**, 14 (1985).
13. Westerman, P. C., *et al.*, "Swine Lagoon Effluent Applied to 'Coastal' Bermuda Grass: III, Irrigation and Rainfall Runoff." *J. Environ. Qual.*, **14**, 22 (1985).
14. Nagpal, N. K., "Long-Term Phosphorus Sorption in a Brunisol in Response to Dosed-Effluent Loading." *J. Environ. Qual.*, **14**, 280 (1985).
15. Thomas, R. E., "Overland Flow: A Decade of Progress." NTIS PB85-233146, U. S. EPA, Washington, D. C. (1984).
16. Smith, R. G., and Schroeder, E. D., "Field studies of the overland flow process for the treatment of raw and primary treated municipal wastewater." *J. Water Pollut. Control Fed.*, **57**, 785 (1985).
17. Abernathy, A. R., *et al.*, "Overland flow wastewater treatment at Easley, S. C." *J. Water Pollut. Control Fed.*, **57**, 291 (1985).
18. Overman, A., and Schanze, T., "Overland Flow Treatment of Wastewater in Florida." EPA-600/2-84-163, NTIS PB85-115798, U. S. EPA, Cincinnati, Ohio (1985).
19. Borrelli, J., *et al.*, "Overland Flow Treatment of Domestic Wastewater in Northern Climates." EPA-600/2-84-161, NTIS PB85-115806, U. S. EPA, Ada, Okla. (1984).
20. Jenkins, T. F., *et al.*, "Toxic Organics Removal Kinetics in Overland Flow Land Treatment." *Water Res. (G. B.)*, **19**, 707 (1985).
21. Crites, R. W., "Nitrogen Removal in Rapid Infiltration Systems." *J. Environ. Eng. Div., Proc. Am. Soc. Civ. Eng.*, **111**, 865 (1985).
22. Bennett, E. R., and Leach, L. E., "Rapid Infiltration Wastewater Treatment for Small Communities." NTIS PB85-238533, U. S. EPA, Cincinnati, Ohio (1985).
23. Reed, S. C., *et al.*, "Problems with rapid infiltration—a post mortem analysis." *J. Water Pollut. Control Fed.*, **57**, 854 (1985).
24. Mansell, R. S., *et al.*, "Phosphate Movement in Columns of Sandy Soil from a Wastewater—Irrigated Site." *Soil Science*, **140**, 59 (1985).
25. Mansell, R. S., *et al.*, "Simulated Solute Movement in Wastewater—Ponded Soil." *Soil Sci. Soc. Am. J.*, **49**, 541 (1985).
26. Tyler, E. J., *et al.*, "Design and Management of Subsurface Soil Absorption Systems." EPA-600/2-85-070, NTIS PB85-216570, U. S. EPA, Cincinnati, Ohio (1985).
27. Farrell, S. O., "Evaluation of Color Infrared Aerial Surveys of Wastewater Soil Absorption Systems." EPA-600/2-85-038, NTIS PB85-189074, U. S. EPA, Cincinnati, Ohio (1985).
28. Weber, A. S., and Tchobanoglous, G., "Rational design for ammonia conversion in water hyacinth treatment systems." *J. Water Pollut. Control Fed.*, **57**, 318 (1985).
29. Godfrey, P. J., *et al.* (Eds.), "Ecological Considerations in Wetlands Treatment of Municipal Wastewaters." Van Nostrand Reinhold Co., New York, N. Y. (1985).

Journal WPCF, Volume 58, Number 6

June 1986

Appendix B

Abridged Version of van Beers' (1963) Publication for K_{sat} Calculation Using the Auger-Hole Method

1. INTRODUCTION

The auger-hole method is a rapid, simple and reliable method for measuring hydraulic conductivity of soil below a water table. It is mostly used in connection with the design of drainage systems in waterlogged land and in canal seepage investigations.

The method, originated by DISERENS (1934), was improved by HOOGHOUDT (1936) and later by KIRKHAM (1945, 1948), VAN BAVEL (1948), ERNST (1950), JOHNSON (1952) and KIRKHAM (1955).

The general principle is very simple: a hole is bored into the soil to a certain depth below the water table. When equilibrium is reached with the surrounding groundwater, a part of the water in the hole is removed. The water seeps into the hole again, and the rate at which the water rises in the hole is measured and then converted by a suitable formula to the hydraulic conductivity (k) for the soil.

5. COMPUTING THE HYDRAULIC CONDUCTIVITY FROM THE DATA OF THE MEASUREMENTS

5.1. GRAPHS

The type of graph developed by BOUMANS (1953; VISSER, 1954) is commonly used in the Netherlands.

Recently other graphs as developed by WESTERHOF (ERNST, WESTERHOF, 1953) have also been used in combination with a special method of recording the readings for each interval of time. Using an additional mechanism on the standard, a pencil point is pressed at each interval of time on a slip of paper attached to the steel tape rule. This piece of paper is placed on a graph then and (in combination with a second graph) the k-factor can be computed. However, these graphs have only been prepared for $r = 4.5$ and $S > \frac{1}{2}H$.
ERNST (1950) prepared graphs for $r = 4$ and $r = 6$, both for $S = 0$ and $S > \frac{1}{2}H$. These graphs are the result of relaxation constructions.
The relation between the k-factor and the rate of rise ($\Delta y/\Delta t$) can be expressed as follows:

$$k = C \frac{\Delta y}{\Delta t} \tag{1}$$

The C value, in its turn is a function of y, H, r and S, which function can be read from the graphs.

Instead of computing the values of k for each Δy_t, these measurements may be averaged before evaluating C from the graphs, provided $\Sigma \Delta y_t < 1/4\ y_o$ and the consecutive readings are reasonably consistent.

Graph 1 ($S \geqslant \frac{1}{2}H$) and *graph* 2 ($S = 0$) – included in this bulletin – are for an auger-hole with a radius of *4 cm*. These graphs are the same as those prepared by ERNST (1950), except that a single logarithmic scale has been used instead of a double one in order to facilitate the reading.

Equation (1) can be solved by means of a nomogram as given on the left side of the graphs or by using a slide rule.

If $S < \frac{1}{2}H$ ($S = \infty$ gives about the same results as $S = \frac{1}{2}H$) no special equation or graph is available. Hence an estimate has to be made between the k-value for $S \geqslant \frac{1}{2}H$ and $S = 0$.
The difference between the C-values for $S \geqslant \frac{1}{2}H$ and $S = 0$ decreases with increasing H and if y is small in relation to H.
In general it can be said that the soil layers at a depth greater than 10–15 cm below the bottom of the hole have little influence on the rate of rise of the water in the hole, and the graph $S \geqslant \frac{1}{2}H$ can be used.

The graphs prepared by ERNST can also be used for an auger-hole with a radius other than 4 or 6 cm. Augers are often used, which have a radius of 5 cm or a 4-inch diameter. *Graphs 3 and 4*, included in this bulletin, have therefore been prepared for *r = 5 cm*. This has been done by converting the graphs for r = 4 cm. [1]

The graphs are used as follows:
C is read from the diagrams as a function of y and H. H is found at the abcissa in cm. Using the line with the proper y value, C can be read as the ordinate (see also example in graph 1).

5.2. FORMULAE

A formula is not often used for computing the hydraulic conductivity since convenient graphs are available. Moreover, unlike the formulae the graphs may be used for a wider range of y and H values and they are more accurate. The difference may amount to 20 percent. For the sake of completeness the formulae, which can be used when no graph is available, will be given here.

The following formula has been obtained for homogeneous soil with the impermeable layer at a certain depth, $S \geqslant \frac{1}{2}H$, below the bottom of the auger-hole (ERNST, 1950).

$$k = \frac{4000}{\left(\frac{H}{r} + 20\right)\left(2 - \frac{y}{H}\right)} \frac{r}{y} \frac{\Delta y}{\Delta t} \qquad (2)$$

In this formula k is expressed in m/24 hours. All other quantities are in cm or in sec.
k = hydraulic conductivity.
H = depth of hole below the groundwater table.
y = distance between groundwater level and the average level of the water in the hole for the time interval Δt.
r = radius of auger-hole.
S = depth of the impermeable layer below the bottom of the hole or the layer, which has a permeability of about one tenth or less of the permeability of the layers above.

[1]) The easiest way of making this conversion is to multiply the values of H and y on the graph r = 4 by 5/4 or 1.25.
For instance: H_{20} on r = 4 = 5/4 × 20 = H_{25} on r = 5,
H_{30} on r = 4 = 5/4 × 30 = $H_{37.5}$ on r = 5.
H_{40} on r = 4 = 5/4 × 40 = H_{50} on r = 5, etc.

The same applies to y.
To facilitate reading the values of H and especially of y, a new graph with regular intervals of y = 20, 25, 30, 35, etc., was constructed.
The values on the graph r = 5 for y = 20, 25, 30, 35, 40, 50, etc., are the same as the y-values 16, 20,

Equation (2) represents an empirically derived approximate expression of the results of a number of relaxation constructions. Hence this formula does not show the exact relationship that should theoretically exist between the different quantities. although the value of k will be sufficiently accurate (maximum error: 20 %) if the following conditions are met:

$$r > 3 \quad \text{and} < 7 \text{ cm;}$$
$$H > 20 \quad \text{and} < 200 \text{ cm;}$$
$$y > 0.2\ H;$$
$$S > H;$$
$$\Delta y \leqslant 1/4\ y_o.$$

Equation (2) can also be written in another form, which facilitates calculations:

$$k = \frac{4000\ r^2}{(H + 20r)\left(2 - \frac{y}{H}\right)y} \cdot \frac{\Delta y}{\Delta t} \tag{2a}$$

When *the impermeable layer is at the bottom of the hole* (S = 0) the following equation can be used:

$$k' = \frac{3600\ r^2}{(H + 10r)\left(2 - \frac{y}{H}\right)y} \cdot \frac{\Delta y}{\Delta t} \tag{3}$$

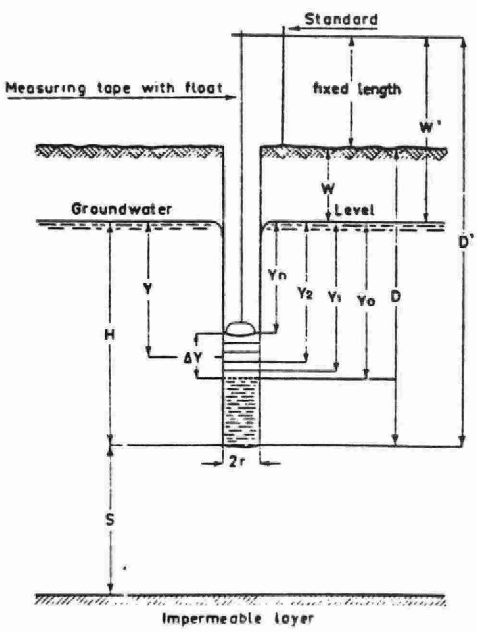
5.3. EXAMPLES

Example No. I demonstrates the computation of the k-factor from the data obtained in the field.

Example No. 2 illustrates some of the irregularities that may occur in the measurement data.

Example No. 3 shows that the k-factor computed will be too low if the measurements are continued for too long ($\Delta y > 1/4\ y_o$) or if too long a period elapses between bailing and starting the measurements.

Fig. 6.



EXAMPLE NO. 1 (FIG. 6)

Note: All readings are taken 40 cm above ground-level (see fig. 4 and fig. 6).

No.:	**Date:**
Location:	**Technician:**

D′ = *240*	**D** = 200	**r** = *4*	**k estm.** = **m/day**
W′ = *114*	**W** = 74	**S** = > *½H*	**k calc.** = 0.66 **m/day**
H = 126	**H** = 126		

t	y_t'	Δy_t
0	*145.2*	
10	144.0	1.2
20	142.8	1.2
30	141.7	1.1
40	140.6	1.1
50	139.6	1.1
$\Delta y =$	5.6	5.6

$y_o = y_o' - W' = 145.2 - 114 = 31.2$

$\Delta y = y_o' - y_n' = \Sigma\Delta y_t = 5.6$

$y = y_o - \frac{1}{2}\Delta y = 31.2 - 2.8 = 28.4$

$\left.\begin{array}{l} H = 126. \\ y = 28.4 \end{array}\right\}$ C = 6.0 (read from Graph 1)

$$\frac{\Delta y}{\Delta t} = \frac{5.6}{50} = 0.11$$

$$k = C\frac{\Delta y}{\Delta t} = 6.0 \times 0.11 = 0.66$$

Note: The text printed on the field-sheet is shown here in **bold** type; the field notes are in *italics*, and the work carried out in the office is designated by roman type.

After the readings have been taken it is desirable to have some check on the reliability of the measurements. The Δy_t of each measurement is, therefore, computed in order to see whether the consecutive readings are reasonable consistent. If the value of Δy_t decreases gradually. the readings may be averaged up to $\Delta y = 1/4\ y_o$ or, in this case, up to a Δy of 7 to 8 cm. Both conditions are met here, so that k can be computed.

t	y_t	Δy_t
0	31.5	
10	*30.0*	- 1.5
20	28.8	- 1.2
30	27.7	- 1.1
40	26.5	- 1.2
50	25.6	- 0.9
60	*24.5*	- 1.1

EXAMPLE NO. 2.

In this example we see that the first Δy_t is somewhat high. This is frequently the case, and is most probably caused by water dropping along the wall of the hole after the water has been bailed out.

Moreover, it is very easy to make a 1 mm error in reading the y_t. The rise of the measuring tape may also be somewhat irregular at times, since the steel tape or float may stick to the wall of the hole. But these errors are sufficiently eliminated by taking the average of 4–6 measurements. In the example given one would take for Δy: 30.0–24.5 = 5.5 and $y = y_o - \frac{1}{2}\Delta y = 30.0 - 2.8 = 27.2$.

For $\Delta t = 50$, $H = 80$, $r = 4$ and $S > H$:

$c = 9.0$ and $k = 9.0 \times 5.5/50 = 0.99$. (Graph 1)

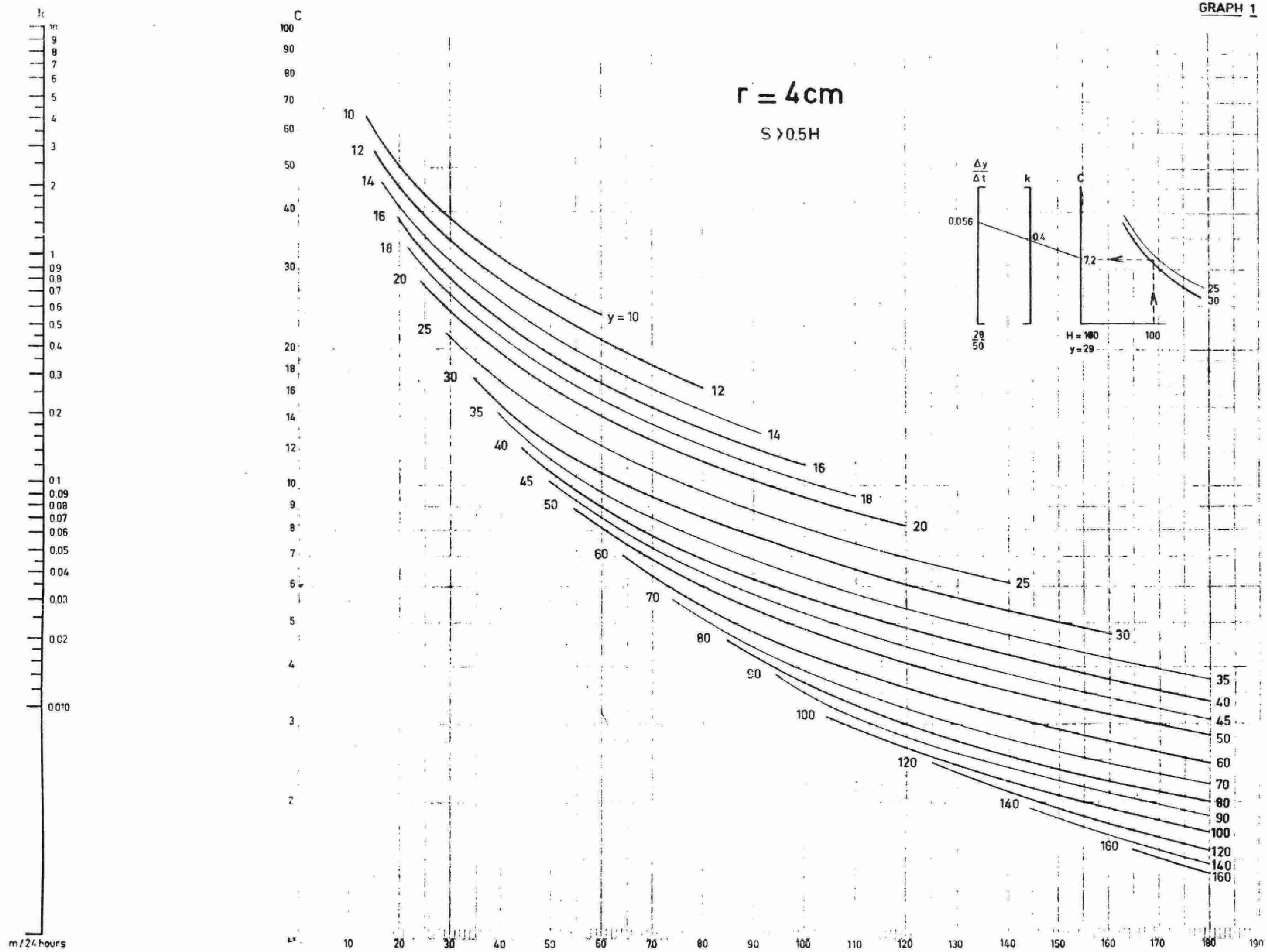
EXAMPLE NO. 3

t	y_t	Δy_t		
0	*30.3*			
15	28.8	− 1.5	$\Delta y/\Delta t = 5.8/60$	r = 4
30	27.3	− 1.5	y = 27.4	$S > \frac{1}{2}H$ graph 1
45	25.9	− 1.4	C = 17.8	H = 35
60	*24.5*	− 1.4	k = *1.75*	
75	23.2	− 1.3	$\Delta y/\Delta t = 4.9/60$	
90	21.9	− 1.3	y = 22.0	
105	20.7	− 1.2	C = 20.2	
120	*19.6*	− 1.1	k = *1.65*	
135	18.6	− 1.0	$\Delta y/\Delta t = 3.8/60$	
150	17.6	− 1.0	y = 17.7	
165	16.7	− 0.9	C = 23.2	
180	*15.8*	− 0.9	k = *1.47*	
195	15.0	− 0.8	$\Delta y/\Delta t = 2.9/60$	
210	14.2	− 0.8	y = 14.3	
225	13.5	− 0.7	C = 27.5	
240	12.7	− 0.7	k = *1.33*	

In order to demonstrate which errors are introduced if the measurements are continued for too long a time the k-factor has been computed for each group of five readings. Although $\Delta y/\Delta t$ should decrease, the product $C\frac{\Delta y}{\Delta t}$, and hence k, should be constant, but it can be seen that this is not the case here.

The decrease in the calculated k-factor is caused by the funnelshaped drawdown of the water-table around the hole and the corresponding decrease in the H-value. In example No. 3 all C-values have been calculated for H = 35 cm, but in this case the apparent H-values will have been H_{35}, H_{32}, H_{28} and H_{24}, respectively. If these H-values had been used, each of the four calculations would have given the same k-factor (k = 1.75).

GRAPH 1



r = 4cm
S > 0.5H
k
C
m/24 hours
y = 10
$\frac{\Delta y}{\Delta t}$
0.056
0.4
7.2
H = 100
y = 29
100

REFERENCES

BAVEL, C. H. M. VAN AND D. KIRKHAM. 1948: Field measurements of soil permeability using auger holes. Proc. Soil. Sc. Soc. Am., pp. 90–96.

BOUMANS, J. H. 1953: Het bepalen van de drainafstand met behulp van de Boorgaten methode. Landbouwk. Tijdschrift, 82–104.

DISERENS, E. 1934: Beitrag zur Bestimmung der Durchlässigkeit des Bodens in natürlicher Bodenlagerung. Schweiz. Landw. Monatstr. 12.

ERNST, L. F. 1950: A new formula for the calculation of the permeability factor with the auger hole method. T.N.O. Groningen 1950. Translated from the Dutch by H. Bouwer, Cornell Univ. Ithaca. N.Y. 1955.

ERNST, L. F. and J. J. WESTERHOF. 1956: Le developpement de la recherche hydrologique et son application au drainage aux Pays-Bas. Assoc. Intern. Hydr., Publ. 41. Symposia Darcy (Dijon, 1956).

HOOGHOUDT, S. B. 1936: Bepaling van den doorlaatfactor van den grond met behulp van pompproeven. (z.g. boorgatenmethode). Verslag Landbouwk. Onderzoek 42, pp. 449–541.

JOHNSON, H. P., R. K. FREVERT, and D. D. EVANS. 1952: Simplified procedure for the measurement and computation of soil permeability below the water-table. Agric. Eng., pp. 283–286.

KIRKHAM, D. 1945: Proposed methods for field measurements of permeability of soil below the water-table. Proc. Soil. Sc. Soc. Am., pp. 58–68.

KIRKHAM, D. AND C. H. M. VAN BAVEL. 1948: Theory of seepage into auger holes. Proc. Soil Sc. Soc. Am., pp. 75–82.

KIRKHAM, D. 1955: Measurement of the hydraulic conductivity of soil in place. Symposium on permeability of soil. Am. Soc. for Testing Materials. Spec. Techn. Publ. No. 163.

VISSER, W. C. 1954: Tile drainage in the Netherlands. Neth. Journal of Agr. Sci. pp. 69 87.

Appendix C

Empirical Equations for Potential Evapotranspiration Estimation Using the Modified Energy Budget Approach

MODIFIED ENERGY APPROACH TO Ep ESTIMATION

DEG = Julian Date + 280
    if DEG > 360, then DEG = DEG - 360

SINFIT = 1.0 + sin(DEG) ← if sine function operates on degrees
       = 1.0 + sin (DEG•0.017453290) ← if sine function operates on radians

$$\mathrm{DLSS} = 16.44736 - (0.17264 \cdot \mathrm{NLAT}) + (0.00343 \cdot [\mathrm{NLAT}]^2)$$
$$\mathrm{DLWS} = 8.73214 + (0.12824 \cdot \mathrm{NLAT}) - (0.00283 \cdot [\mathrm{NLAT}]^2)$$
$$\mathrm{RADSS} = 35.47561 + (0.33508 \cdot \mathrm{NLAT}) - (0.00382 \cdot [\mathrm{NLAT}]^2)$$
$$\mathrm{RADWS} = 44.83720 - (0.90160 \cdot \mathrm{NLAT}) + (0.00312 \cdot [\mathrm{NLAT}]^2)$$

$$\mathrm{SUNPOS} = \left[\mathrm{SINFIT} \cdot \left[\frac{(\mathrm{DLSS} - \mathrm{DLWS})}{2}\right]\right] + \mathrm{DLWS}$$

$$\mathrm{RADPOS} = \left[\mathrm{SINFIT} \cdot \left[\frac{(\mathrm{RADSS} - \mathrm{RADWS})}{2}\right]\right] + \mathrm{RADWS}$$

$$\mathrm{RADOBS} = \mathrm{RADPOS} \cdot \left(0.23 + \left[0.57 \cdot \frac{\mathrm{SUNOBS}}{\mathrm{SUNPOS}}\right]\right)$$

$$\mathrm{RADNET} = (\mathrm{RADOBS} \cdot 0.54) - 0.754$$

$$\mathrm{TDMEAN} = (0.75 \cdot \mathrm{TMAX}) + (0.25 \cdot \mathrm{TMIN})$$

$$\mathrm{PEVSRN} = 0.516 + (0.02 \cdot \mathrm{TDMEAN}) - (0.000152 \cdot [\mathrm{TDMEAN}]^2)$$
    if PEVSRN > 1, then PEVSRN = 1
    if PEVSRN < 0, then PEVSRN = 0

$$\mathrm{PE} = \mathrm{PEVSRN} \cdot (\mathrm{RADNET} \cdot 0.4049)$$
    if PE < 0, then PE = 0

JULIAN DATE = the number of days from January 1 (e.g. April 1 = day 91)
NLAT = degrees orth latitude (e.g. 45°05' = 45.083)
DLSS = duration of daylight at the summer solstice (hours)
RADSS= daily solar radiation at the top of the atmosphere at the summer solstice ($MJ \cdot m^{-2} \cdot d^{-1}$)
DLWS= duration of daylight at the winter solstice (hours)
RADWS= daily solar radiation at the top of the atmosphere at the winter solstice ($MJ \cdot m^{-2} \cdot d^{-1}$)
SUNPOS= photoperiod or daily duration of daylight (hours)
RADPOS= daily solar radiation at the top of the atmosphere ($MJ \cdot m^{-2} \cdot d^{-1}$)
SUNOBS= daily duration of bright sunshine (hours)
RADOBS= daily global solar radiation ($MJ \cdot m^{-2} \cdot d^{-1}$)
RADNET= daily net radiation ($MJ \cdot m^{-2} \cdot d^{-1}$)
TDMEAN= mean daytime temperature (°C)
PEVSRN= potential evapotranspiration vs. net radiation relationship for a specified daytime mean temperature (i.e. slope of the saturation vapor pressure - temperature curve)
PE= potential evapotranspiration (mm)

Appendix D

Tables of Morphological, Physical and Chemical Soil Characterization Data

[redacted]e D[redacted]Ped[redacted]ica[redacted]il [redacted]file[redacted]scr[redacted]ons[redacted] si[redacted]mp[redacted] lo[redacted]ons.[redacted]

Soil Pit No.	Leachate Spray Regime	Soil Horizon	Depth	Textural Class	Drainage Class	Munsell Colour
			cm			
1	none	LFH/Ah	0-7	-	Imperfect	10YR 2/1(black)
		Ahe	7-11	vfsl		10YR 3/2 (very dark grayish brown)
		Bhfj	11-25	lfs		5YR 3/3 (dark reddish brown)
		BCgj	25-42	lfs		10YR 4/4 (dark yellowish brown)
		Cgj	42+	lfs		10YR 5/4 (yellowish brown)
2	none (station 1)	LFH/Ah	0-2	-	Excessively good	10YR 2/1 (black)
		Ahe	2-7	lfs		10YR 3/2 (very dark grayish brown)
		Bhfj	7-18	lfs		2.5YR 3/2 (dusky red)
		Bm	18-30	s		5YR 3/3 (dark reddish brown)
		BC	30-65	lfs		10YR 5/4 (yellowish brown)
		Cc	65+	fs		10YR 6/2 (light brownish gray)
		C	65+	s		10YR 6/2 (light brownish gray)
3	light (6m from nozzle)	LFH/Ah	0-7	-	Excessively good	10YR 2/1 (black)
		Ahe	7-13	ls		10YR 3/2 (very dark grayish brown)
		Bhfc	13-29	s		5YR 3/4 (dark reddish brown)
		Bm	29-49	fs		10YR 5/6 (yellowish brown)
		BC_1	49-65	fs		10YR 5/4 (yellowish brown)
		BC_2	65-80	fs		10YR 5/4 (yellowish brown)
		C	80+	fs		10YR 6/2 (light brownish gray)
4	light (1m from nozzle at station 2)	LFH/Ah	0-14	ls	Excessively good	10YR 2/1 (black)
		Ahe	14-22	s		10YR 3/2 (very dark grayish brown)
		Bhf	22-55	s		10YR 2.5/1 (reddish black)
		Bfc	55-84	s		10YR 3/4 (dusky red)
		Bfj	84-115	s		5YR 3/4 (dark reddish brown)
		BC	115-160	s		10YR 5/4 (yellowish brown)
		C	160+	s		10YR 6/2 (light brownish gray)
5	heavy (3m from nozzle at station 3)	LFH/Ah	0-18	lfs	Imperfect to good	10YR 2/1 (black)
		Ahe	18-24	fs		10YR 3/1 (very dark gray)
		Bhfc (humic)	24-50	fs		5YR 2.5/1 (black)
		Bhfc (ferric)	24-50	fs		2.5YR 3/2 (dusky red)
		Bfc	50-gwt.	lfs		5YR 3/3 (dark reddish brown)
6	none (control permanent plot)	LFH/Ah	0-4	fsl	Excessively good	10YR 2/1 (black)
		Ahe	4-7	-		10YR 3/1 (very dark gray)
		Bhfj	7-13	fsl		7.5YR 3/2 (dark brown)
		Bm	13-48	fsl		7.5YR 3/2 (dark brown)
		C	48+	vglcs		10YR 3/3 (dark brown)

Table D.2. Soil physical characterization for six sampling locations.

Soil Pit No.	Soil Horizon	Depth	Dry Bulk Density	Organic Matter Content	Textural Class	Gravel	Particle Size Distribution vcs	cs	ms	fs	vfs	total sand	total silt	total clay
		cm	$g \cdot cm^{-3}$	% $g \cdot g^{-1}$		% $g \cdot g^{-1}$	% $g \cdot g^{-1}$							
1	LFH/Ah	0-7	0.31	15.1	-	-	-	-	-	-	-	-	-	-
	Ahe	7-11	0.91	5.0	vfsl	0.1	0.5	2.2	6.4	26.7	32.2	68.11	26.4	5.5
	Bhfj	11-25	0.60	6.7	lfs	0.3	3.7	3.0	6.1	28.8	36.1	77.65	20.2	2.1
	BCgj	25-42	1.15	3.2	lfs	1.1	1.6	2.2	4.7	28.0	37.5	73.92	25.7	0.4
	Cgj	42+	1.23	2.8	lfs	0.6	1.1	1.7	4.6	28.2	35.5	71.15	28.9	0.0
2	LFH/Ah	0-2	-	-	-	-	-	-	-	-	-	-	-	-
	Ahe	2-7	1.06	4.0	lfs	0.8	0.8	4.2	15.6	33.4	22.0	76.06	19.0	5.0
	Bhfj	7-18	0.86	5.3	lfs	2.0	1.0	4.7	16.7	34.7	23.6	80.72	17.2	2.1
	Bm	18-30	0.95	5.7	s	9.0	1.8	6.8	21.6	35.1	21.1	86.31	12.9	0.8
	BC	30-65	1.35	2.9	lfs	6.8	0.8	3.1	11.9	42.4	25.9	84.07	15.5	0.4
	Cc	65+	1.63	0.2	fs	0.3	0.6	1.8	13.2	46.3	26.9	88.91	11.1	0.0
	C	65+	1.55	0.8	s	1.1	0.4	1.7	37.4	49.5	8.5	97.58	2.4	0.0
3	LFH/Ah	0-7	-	-	-	-	-	-	-	-	-	-	-	-
	Ahe	7-13	1.42	3.0	ls	0.3	0.5	4.0	24.1	40.4	15.1	84.08	12.2	3.7
	Bhfc	13-29	1.07	5.6	s	18.2	1.8	4.3	20.1	46.0	17.5	89.65	9.5	0.8
	Bm	29-49	1.16	3.0	fs	1.3	0.8	2.4	18.4	48.7	21.5	91.83	8.2	0.0
	BC_1	49-65	1.41	2.0	fs	1.0	0.6	2.8	19.0	48.7	20.3	91.42	8.6	0.0
	BC_2	65-80	1.54	1.5	fs	2.5	1.4	3.5	23.2	51.6	17.1	96.74	3.3	0.0
	C	80+	1.53	2.0	fs	0.0	0.6	2.3	24.0	58.4	13.0	99.19	0.4	0.4
4	LFH/Ah	0-14	0.85	6.3	ls	1.3	1.2	4.4	29.1	39.8	9.4	83.96	8.9	7.2
	Ahe	14-22	1.27	2.3	s	3.4	0.6	4.5	32.8	47.4	9.5	94.71	4.1	1.2
	Bhf	22-55	1.16	3.9	s	5.2	0.6	3.5	32.4	46.1	9.9	92.57	5.0	2.5
	Bfc	55-84	1.32	3.2	s	0.4	1.6	10.3	42.6	38.3	5.5	98.36	1.2	0.4
	Bfj	84-115	1.53	1.8	s	3.5	4.7	14.0	40.3	35.1	5.6	99.59	0.4	0.0
	BC	115-160	1.57	1.2	s	0.1	0.3	6.7	36.4	46.9	8.9	99.19	0.8	0.0
	C	160+	1.66	0.2	s	19.0	4.0	10.0	47.8	34.3	3.5	99.60	0.4	0.0
5	LFH/Ah	0-18	-	6.5	lfs	0.3	1.2	2.3	16.7	50.7	12.1	83.02	9.8	7.2
	Ahe	18-24	-	3.6	fs	0.3	0.9	2.1	15.7	54.2	14.4	87.25	9.9	2.9
	(humic) Bhfc	24-50	-	6.2	fs	0.0	0.2	1.8	17.7	55.7	12.4	87.80	7.2	5.0
	(ferric) Bhfc	24-50	-	5.9	fs	0.8	0.7	2.3	19.3	56.3	12.9	91.55	5.5	3.0
	Bfc	50-gwt.	-	5.4	lfs	0.0	0.4	2.7	20.7	50.2	11.0	85.11	8.7	6.2
6	LFH/Ah	0-4	-	8.1	fsl	4.8	4.1	8.2	12.5	18.5	13.5	56.83	32.4	10.8
	Ahe	4-7	-	-	-	-	-	-	-	-	-	-	-	-
	Bhfj	7-13	-	7.1	fsl	8.6	4.2	8.7	13.4	19.5	20.9	66.64	30.8	2.6
	Bm	13-48	-	5.7	fsl	9.1	4.8	7.9	12.2	18.0	16.1	58.97	40.2	0.8
	C	48+	-	2.3	vglcs	58.5	10.8	20.7	23.3	18.7	9.3	82.86	17.1	0.0

Table D.3. Soil moisture desorption data for four sampling locations.

Soil Pit No.	Soil Horizon	Depth	Dry Bulk Density	Volumetric Soil Moisture Content at Prescribed Tensions: 0kPa	5 kPa	10 kPa	33 kPa	100 kPa	1.5 MPa	Plant-Available Soil Water
		cm	$g \cdot cm^{-3}$	$m^3 \cdot m^{-3}$						
1	LFH/Ah	0-7	0.29	0.86	0.56	0.51	0.49	0.42	0.39	0.10-0.12
	Ahe	7-11	0.84	0.53	0.43	0.39	0.36	0.32	0.30	0.06-0.09
	Bhfj	11-25	0.82	0.62	0.44	0.36	0.29	0.25	0.25	0.04-0.11
	BCgj	25-42	1.21	0.56	0.36	0.28	0.18	0.14	0.12	0.06-0.16
	Cgj	42+	1.32	0.48	0.39	0.29	0.17	0.13	0.09	0.08-0.20
2	LFH/Ah	0-2	-	-	-	-	-	-	-	-
	Ahe	2-7	1.11	0.62	0.36	0.26	0.22	0.19	0.19	0.03-0.07
	Bhfj	7-18	1.03	0.62	0.43	0.36	0.32	0.23	0.21	0.11-0.15
	Bm	18-30	1.01	0.58	0.41	0.35	0.31	0.22	0.21	0.10-0.14
	BC	30-65	1.27	0.52	0.35	0.27	0.17	0.14	0.12	0.05-0.15
	Cc	65+	1.62	0.37	0.26	0.15	0.04	0.03	0.02	0.02-0.13
	C	65+	1.52	0.46	0.12	0.05	0.04	0.03	0.02	0.02-0.03
3	LFH/Ah	0-7	-	-	-	-	-	-	-	-
	Ahe	7-13	1.50	0.55	0.32	0.26	0.18	0.13	0.10	0.08-0.16
	Bhfc	13-29	1.02	0.60	0.40	0.30	0.23	0.20	0.18	0.05-0.12
	Bm	29-49	1.22	0.58	0.34	0.27	0.18	0.15	0.15	0.03-0.12
	BC_1	49-65	1.39	0.44	0.29	0.16	0.11	0.09	0.08	0.03-0.08
	BC_2	65-80	1.61	0.43	0.16	0.09	0.06	0.05	0.04	0.02-0.05
	C	80+	1.66	0.40	0.16	0.08	0.04	0.03	0.02	0.02-0.06
4	LFH/Ah	0-14	0.85	0.70	0.47	0.40	0.34	0.26	0.17	0.17-0.23
	Ahe	14-22	1.40	0.53	0.24	0.17	0.11	0.10	0.06	0.05-0.11
	Bhf	22-55	1.27	0.55	0.32	0.26	0.20	0.17	0.13	0.07-0.13
	Bfc	55-84	1.35	0.52	0.19	0.15	0.11	0.10	0.09	0.02-0.06
	Bfj	84-115	1.56	0.49	0.13	0.09	0.07	0.06	0.05	0.02-0.04
	BC	115-160	-	-	-	-	-	-	-	-
	C	160+	1.59	0.40	0.11	0.06	0.04	0.03	0.02	0.02-0.04

Table D.4. Soil chemical characterization for six sampling locations.

Soil Pit No.	Soil Horizon	Depth	Elect. Cond.	Cation Exchange Capacity	Kjeldahl N	NO_3^-	P	K	Mg	Ca	Mn	Zn	Fe	Cu	pH	Buffer pH
		cm	msiemens·cm^{-1}	meq.$100g^{-1}$	%g.g^{-1}	----------------------				ppm	----------------------					
1	LFH/Ah	0-7	-	23.40	0.45	-	17	69	96	647	107	58	-	-	4.0	5.0
	Ahe	7-11	0.23	8.50	0.12	15.0	6	23	38	149	13	21	196	0.57	6.8	na.
	Bhfj	11-25	0.16	9.50	0.15	10.0	1	9	26	80	50	40	62	0.30	5.1	5.6
	BCgj	25-42	0.09	2.70	0.06	2.7	2	7	20	15	31	27	17	0.24	5.7	6.5
	Cgj	42+	0.12	1.30	0.03	3.0	3	3	18	35	31	27	17	0.25	5.7	6.8
2	LFH/Ah	0-2	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	Ahe	2-7	0.21	5.30	0.11	3.0	1	28	31	84	82	58	173	0.43	4.4	6.2
	Bhfj	7-18	0.17	6.70	0.10	14.0	1	10	22	7	54	51	104	0.50	5.0	5.6
	Bm	18-30	0.12	5.20	0.09	9.0	1	6	16	1	51	40	46	0.44	5.1	5.8
	BC	30-65	0.09	1.00	0.02	3.0	5	5	18	1	26	24	19	0.20	5.9	6.8
	Cc	65+	0.86	0.03	0.01	11.0	1	3	17	1	22	22	45	0.26	6.1	na.
	C	65+	0.16	0.03	0.01	4.0	15	3	15	20	18	20	31	0.24	6.3	na.
3	LFH/Ah	0-7	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	Ahe	7-13	0.18	3.70	0.07	16.0	12	26	20	63	83	53	162	0.35	4.4	6.2
	Bhfc	13-29	0.22	4.90	0.09	21.0	1	13	17	68	55	42	85	0.35	5.0	5.8
	Bm	29-49	0.20	1.40	0.04	15.0	1	5	18	42	47	34	28	0.22	5.2	6.4
	BC_1	49-65	0.18	0.17	0.03	6.0	1	4	18	121	29	25	24	0.30	5.8	6.7
	BC_2	65-80	0.19	0.03	0.02	4.0	3	3	19	13	27	24	30	0.34	5.9	7.1
	C	80+	0.18	0.03	0.01	3.0	3	1	16	1	17	19	14	0.26	6.4	na.
4	LFH/Ah	0-14	0.38	7.60	0.01	45.0	7	60	69	408	64	59	209	0.58	4.8	6.3
	Ahe	14-22	0.19	1.50	0.05	19.0	8	25	50	198	37	34	176	0.32	5.5	7.1
	Bhf	22-55	0.28	6.20	0.09	29.0	5	37	54	381	41	35	188	0.41	5.4	6.0
	Bfc	55-84	0.14	3.20	0.05	9.0	1	23	24	69	37	29	119	0.36	5.5	6.3
	Bfj	84-115	0.14	0.03	0.02	5.0	16	21	23	16	22	22	23	0.34	6.1	na.
	BC	115-160	0.16	0.03	0.01	7.0	13	27	22	33	22	22	30	0.28	6.1	na.
	C	160+	0.19	0.04	0.01	6.0	10	10	22	1	19	20	27	0.50	6.3	na.
5	LFH/Ah	0-18	0.37	11.10	0.19	43.0	4	87	147	1271	19	87	156	0.54	6.4	na.
	Ahe	18-24	0.28	4.40	0.10	27.0	5	47	91	787	20	25	150	0.42	6.3	na.
	(humic)Bhfc	24-50	0.30	11.50	0.14	31.0	1	69	119	899	23	31	186	0.63	6.1	na.
	(ferric)Bhfc	24-50	0.19	8.40	0.10	14.0	1	46	65	488	34	28	131	0.41	5.6	6.0
	Bfc	50-gwt.	0.20	7.80	0.07	12.0	1	61	80	473	34	37	183	0.39	5.6	6.0
6	LFH/Ah	0-4	0.36	15.0	0.28	42.0	24	40	50	541	125	106	184	1.06	3.7	5.7
	Ahe	4-7	-	-	-	-	-	-	-	-	-	-	-	-	-	-
	Bhfj	7-13	0.16	9.00	0.12	8.0	1	10	14	37	72	49	97	0.61	4.6	5.5
	Bm	13-48	0.13	4.50	0.11	5.0	1	9	17	6	63	41	35	0.41	4.8	5.9
	C	48+	0.17	1.20	0.19	6.0	6	7	18	1	47	34	69	0.42	5.2	6.6

na. - not applicable (i.e. buffer pH determination not routinely performed for soil pH of 6.0 or greater).

Appendix E

Graphs of Measured Soil Moisture Regimes

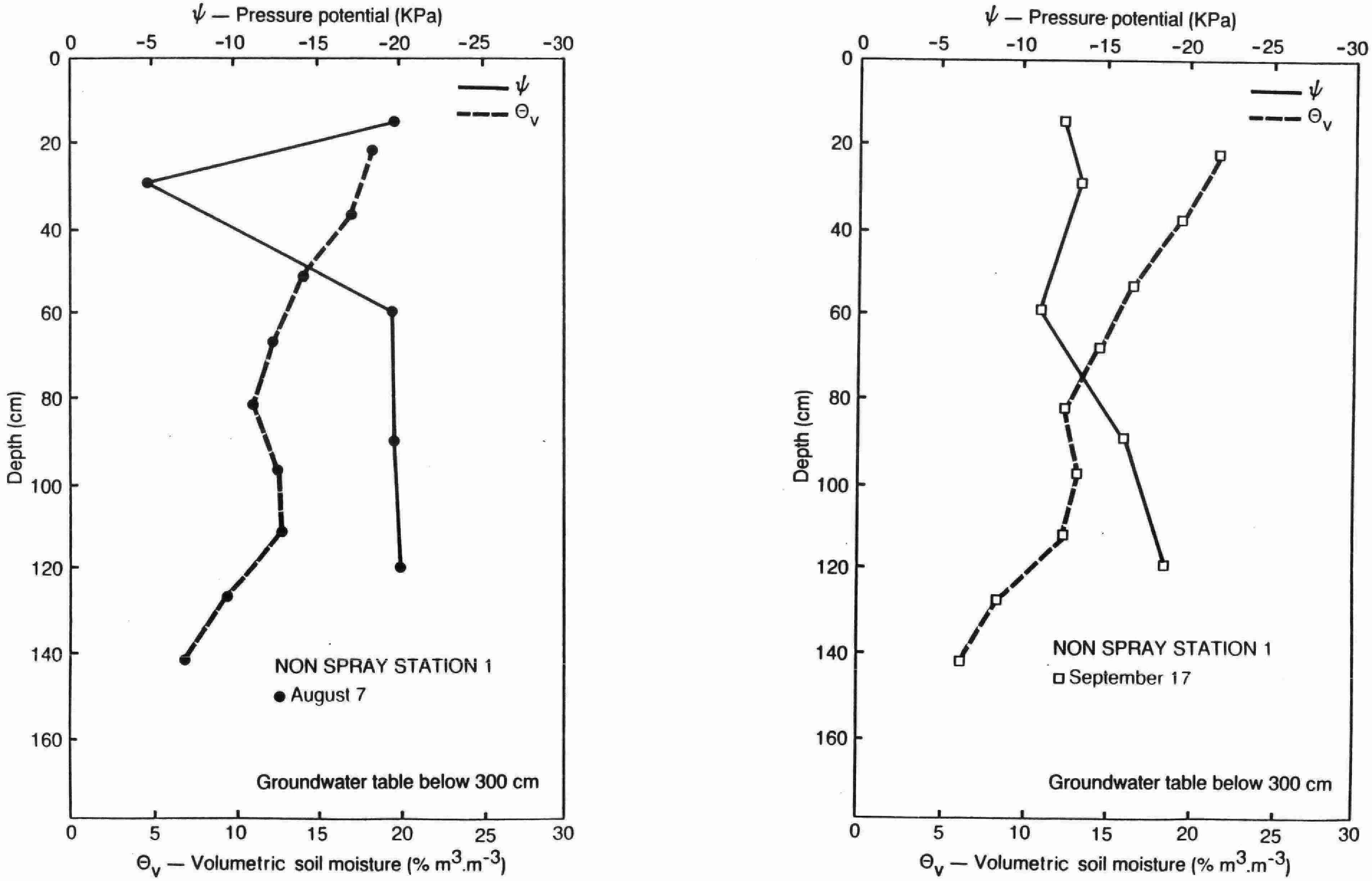


Figure E.1. Soil moisture regime at station 1 (unsprayed area) on a) August 7 and b) September 17, 1986.

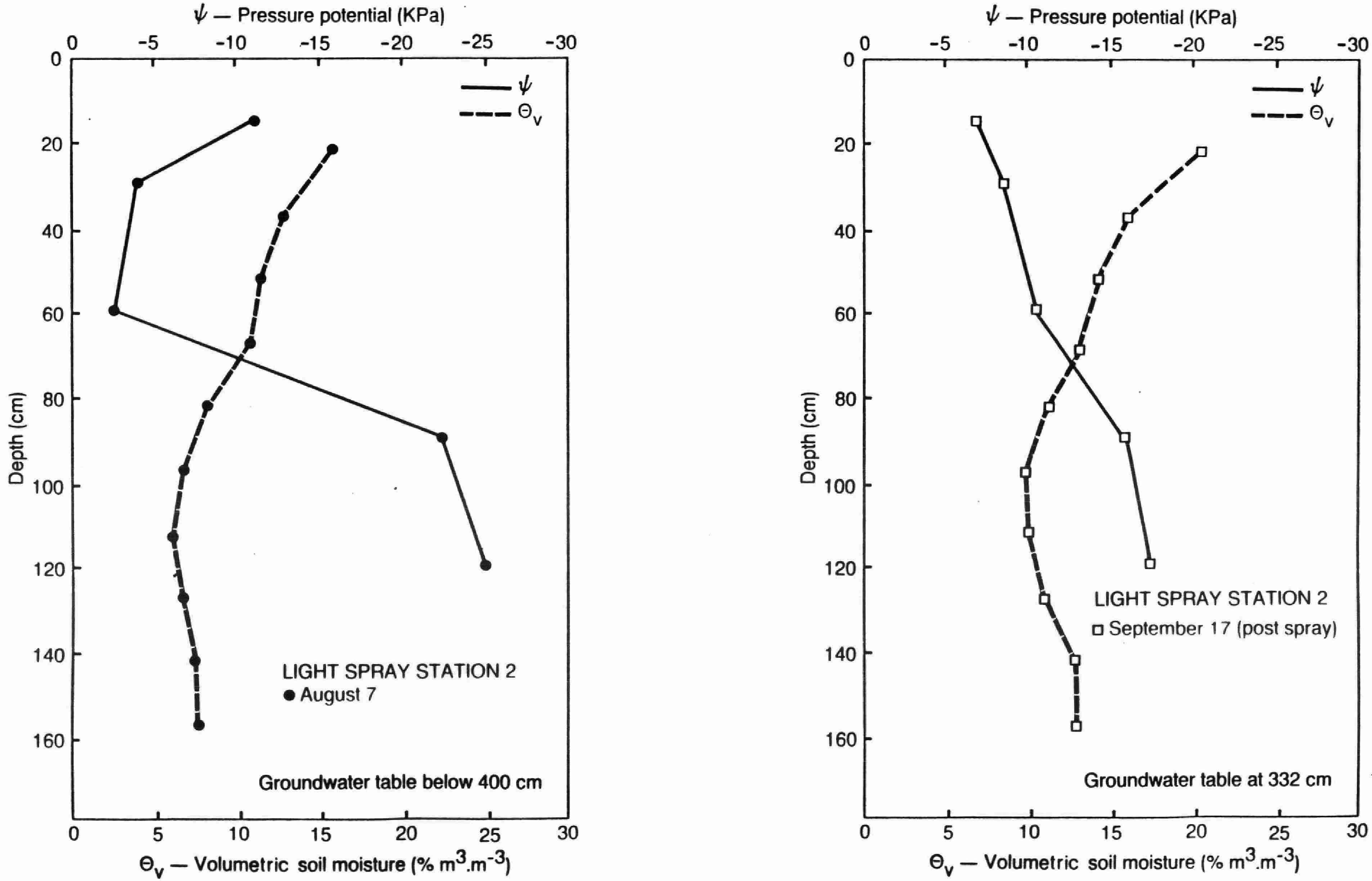
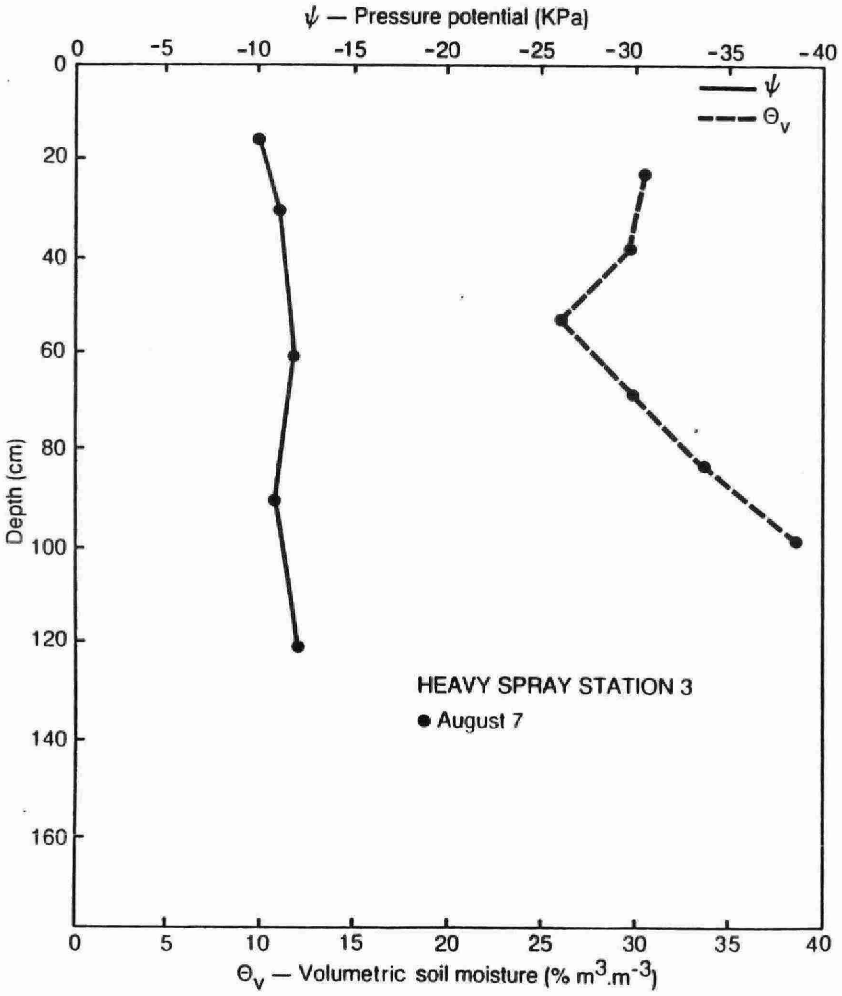


Figure E.2. Soil moisture regime at station 2 (light spray area) on a) August 7 and b) September 17, 1986.



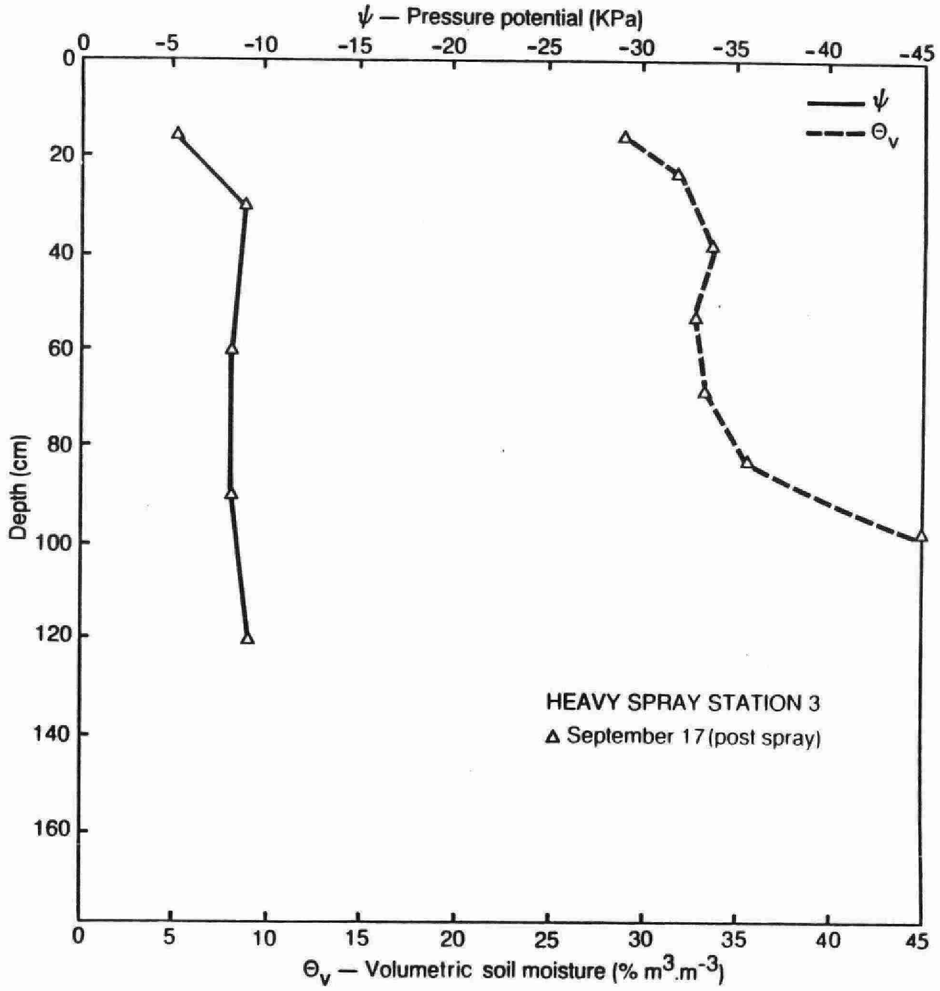


Figure E.3. Soil moisture regime at station 3 (heavy spray area) on a) August 7 and b) September 17, 1986.

Appendix F

Effluent Quality Data from Truncated and Intact Soil Leaching Columns

Table F.1. Range of relative concentrations and attenuation/elution patterns of leachate constituents from leaching columns of soil sprayed in the field (1980-86) and subjected to the moderate application rate (7.0 $mm \cdot d^{-1}$).

Constituent	Untreated		Lime-Treated		Carbon-Treated		Rain Water(Control)	
	Truncated	Intact	Truncated	Intact	Truncated	Intact	Truncated	Intact
$N-NH_4^+$	0.0 A	0.0 A	0.0 A	0.0 A	n.c	n.c.	n.c.	n.c.
$N-NO_3^-$	0.8-330.0 E→A	27-39 E	0.7 A	0.0 A	1.4-5.3 E	0.3-2.5 A→E	3.0-88.0 E	1.25-75.0 E
P	n.c.	n.c.	n.c.	n.c.	n.c.	n.c.	n.c.	n.c.
K	0.7-1.3 E→A	0.8-1.3 E→A	0.5-0.7 A	0.5-0.6 A	0.5-0.7 A	0.0-0.02 A	3.0-11.6 E	3.4-12.3 E
Ca	0.3-1.4 E→A	0.3-0.8 A	0.4-1.8 E→A	0.2-1.0 A	0.3-0.6 A	0.1-0.5 A	0.2-39.3 E→A	0.1-26.7 E→A
Mg	0.8-3.9 E→A	1.0-2.7 E	1.5-25.1 E	5.4-18.8 E	0.7-1.3 E→A	0.1-0.7 A	0.4-36.8 E→A	0.3-45.8 E→A
Mn	0.1-0.4 A	0.1-0.3 A	0.3-38.0 A→E	2.3-7.3 E	12.0-69.0 E	8.0-15.0 E	3.0-18.0 E	0.0-8.0 E→A
Fe	0.0 A	0.0-0.34 A	n.c.	n.c.	n.c.	n.c.	0.0-0.30 A	0.0 A

n.c.- relative concentrations not calculated since influent concentration below detectable limits
A - attenuation
E - elution
E→A - initial elution, subsequent attenuation
A→E - initial attenuation, subsequent elution

Table F.2. Range of relative concentrations and attenuation/elution patterns of leachate constituents from leaching columns of previously unsprayed soil and subjected to the moderate application rate (7.0 $mm \cdot d^{-1}$).

Constituent	Untreated		Lime-Treated		Carbon-Treated		Rain Water (Control)	
	Truncated	Intact	Truncated	Intact	Truncated	Intact	Truncated	Intact
$N\text{-}NH_4^+$	0.0 A	0.0 A	0.0 A	0.0 A	n.c.	n.c.	n.c.	n.c.
$N\text{-}NO_3^-$	0.4-7.5 E→A	1.1-8.0 E	1.5-4.3 E	0.0-5.3 E→A	0.0-5.3 A→E	0.6-2.9 E	0.0-10.0 E→A	1.0-21.0 E→A
P	n.c.	n.c.	n.c.	n.c.	n.c.	n.c.	n.c.	n.c.
K	0.0-0.1 A	0.0-0.1 A	0.01 A	0.0-0.01 A	0.01-0.5 A	0.0-0.02 A	0.1-0.2 A	0.1-0.2 A
Ca	0.1-0.2 A	0.1-0.2 A	0.1-0.4 A	0.1-0.6 A	0.1-0.3 A	0.1-0.3 A	0.1-5.2 E→A	0.1-9.0 E→A
Mg	0.1-0.2 A	0.1-0.3 A	0.4-3.2 E→A	0.4-3.2 E→A	0.1-0.7 A	0.1-0.3 A	0.1-3.7 E→A	0.1-6.9 E→A
Mn	0.2 A	0.0-0.02 A	0.0-5.0 E	0.3-16.4 A→E	4.0-12.0 E	4.4-8.0 E	1.3-2.5 E	0.0-2.5 E→A
Fe	0.0-0.11 A	0.11-0.22 A	n.c.	n.c.	n.c.	n.c.	0.0-0.02 A	0.0 A

n.c. - relative concentration not calculated since influent concentration below detectable limits
A - attenuation
E - elution
E→A - initial elution, subsequent attenuation
A→E - initial attenuation, subsequent elution

Appendix G

Understory Species Classification Using the Mueller-Dombois and Ellenberg (1974) System

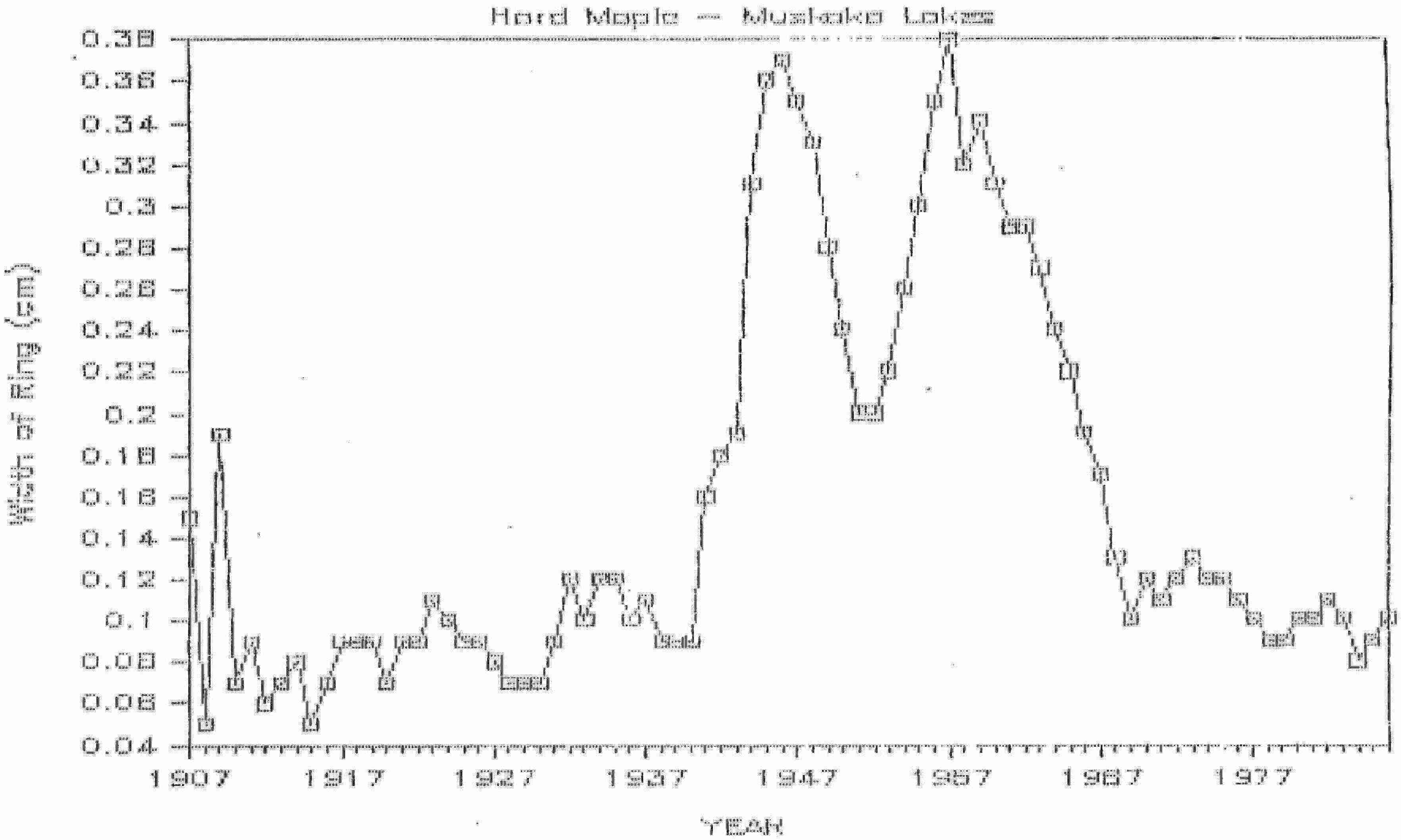
Table G.1. Results of Mueller-Dombois and Ellenberg Classification for the understory vegetation at the Muskoka Lakes site.

Veg. Unit	Vegetation Species	Plot Spray Regime: Heavy			Light			Control			Total Species Count
S	Dryopteris austriaca (Shield Fern)				P	P	P				3
P	*Allium tricoccum (Wild Leek)	P	P				P				3
R	Oryzopsis racemosa (Mountain-rice)	P	P	P			P			P	5
A	*Carex aquatilis (Sedge species)	P	P			P	P			P	5
Y											
C O	Dryopteris spinulosa var. intermedia (Spinulose Wood-Fern)							P	P	P	3
N	Salix species (Willow)							P		P	2
T	Streptopus amplexifolus (Twisted stalk)							P		P	2
R	Hepatica acutiloba (Hepatica)						P		P	P	3
O	*Viburnum alnifolium (Hobble-bush)				P			P	P		3
L											
U	Fagus grandifolia (American Beech)		P	P	P	P	P	P	P	P	8
N	Acer rubrum (Red Maple)	P	P	P	P		P	P	P	P	8
D	*Fraxinus nigra (Black Ash)	P	P			P	P	P	P	P	7
E	Acer pennsylvanicum (Stripled Maple)	P			P	P	P	P		P	6
F	Carex pennsylvanica (Sedge species)		P	P			P		P	P	5
I	*Thelypteris palustris (Marsh Fern)		P	P							2
N	Rubus allegheniesis (Common Blackberry)	P		P							2
E	Galium lanceoaltum (Bedstraw)			P			P				2
D	Trillium grandiflorum (Trillium)					P			P		2
	*Solidago rugosa (Rough Stemmed Goldenrod)		P								1
	*Carex vulpinoidea (Sedge species)	P									1
	Polygonum cilinode (Bindweed)	P									1
	*Impatiens biflora (Jewel-weed)	P									1
	*Geranium Robertianum (Herb Robert)			P							1
	Quercus rubra (Red Oak)					P					1
	Ostrya virgiana (Hop-hornbeam)					P					1
	*Lonicera oblongifolia (Fly Honeysuckle)								P		1
	Osmorhiza longistillis (Sweet Cicely)									P	1
	Epifagus virgiana (Beech Drops)									P	1
	*Viola cucullata (Blue Marsh-violet)									P	1
	Carex plantaginea (Sedge species)									P	1
	*Rhamnus alnifolium (Alder-leaved Buckthorn)			P							1
	Oryzopsis asperifolia (Winter Grass)								P		1
	Number of species per plot	10	8	10	5	8	11	8	10	15	
	Average number of species per treatment area		17			15			19		
	Number of species preferring wet habitats	5	3	3	1	2	3	2	3	1	
	Total number of species preferring wet habitats per treatment area (a total of 12 were found)		9/12			4/12			5/12		

P - present (observed)
* - hydrophyllic species, as identified by at least 4 out of 6 of the following references: Dore and McNeill (1980), Gleason and Conquist (1963), Fernald (1970), Soper and Heimburger (1982), Voss (1972; 1985).

Appendix H

Example Output from the T.R.I.M. System



Hard Maple — Muskoka Lakes
Width of Ring (cm)
0.38
0.36
0.34
0.32
0.3
0.28
0.26
0.24
0.22
0.2
0.18
0.16
0.14
0.12
0.1
0.08
0.06
0.04
1907
1917
1927
1937
1947
1957
1967
1977
YEAR